Northeast Fisheries Science Center Reference Document 10-17

50th Northeast Regional Stock Assessment Workshop (50th SAW):

Assessment Report

by Northeast Fisheries Science Center

August 2010

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10-11 *Northeast Fisheries Science Center Cetacean Biopsy Training Manual*, by F Wenzel, J Nicolas, F Larsen, and RM Pace III. July 2010.

10-12 *A Survey of Social Capital and Attitudes toward Management in the New England Groundfish Fishery*, by DS Holland, P Pinto da Silva, and J Wiersma. July 2010.

10-13 *Black Sea Bass 2010 Stock Assessment Update*, by GR Shepherd and J Nieland. July 2010.

10-14 *Stock Assessment of Summer Flounder for 2010*, by M Terceiro. July 2010.

10-15 *Bluefish 2010 Stock Assessment Update*, by GR Shepherd and J Nieland. July 2010.

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by Northeast Fisheries Science Center

NOAA, National Marine Fisheries Service, 166 Water Street, Woods Hole MA 02543

US DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Northeast Fisheries Science Center
Woods Hole, Massachusetts

August 2010

Northeast Fisheries Science Center Reference Documents

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Editorial Treatment: To distribute this report quickly, it has not undergone the normal technical and copy editing by the Northeast Fisheries Science Center's (NEFSC's) Editorial Office as have most other issues in the NOAA Technical Memorandum NMFS-NE series. Other than the four covers and first two preliminary pages, all writing and editing have been performed by the authors listed within. This report was reviewed by the Stock Assessment Review Committee, a panel of assessment experts from the Center for Independent Experts (CIE), University of Miami.

Information Quality Act Compliance: In accordance with section 515 of Public Law 106-554, the Northeast Fisheries Science Center completed both technical and policy reviews for this report. These predissemination reviews are on file at the NEFSC Editorial Office.

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Foreword

The Northeast Regional Stock Assessment Workshop (SAW) process has three parts: preparation of stock assessments by the SAW Working Groups and/or by ASMFC Technical Committees / Assessment Committees; peer review of the assessments by a panel of outside experts who judge the adequacy of the assessment as a basis for providing scientific advice to managers; and a presentation of the results and reports to the Region's fishery management bodies.

Starting with SAW-39 (June 2004), the process was revised in two fundamental ways. First, the Stock Assessment Review Committee (SARC) became smaller panel with panelists provided by the Independent System for Peer Review (Center of Independent Experts, CIE). Second, the SARC provides little management advice. Instead, Council and Commission teams (e.g., Plan Development Teams, Monitoring and Technical Committees, Science and Statistical Committee) formulate management advice, after an assessment has been accepted by the SARC. Starting with SAW-45 (June 2007) the SARC chairs were from external agencies, but not from the CIE. Starting with SAW-48 (June 2009), SARC chairs are from the Fishery Management Council's Science and Statistics Committee (SSC), and not from the CIE. Also at this time, some assessment Terms of Reference were revised to provide additional science support to the SSCs, as the SSC's are required to make annual ABC recommendations to the fishery management councils.

Reports that are produced following SAW/SARC meetings include: An *Assessment Summary Report* - a summary of the assessment results in a format useful to managers; an *Assessment Report* – a detailed account of the assessments for each stock; and the SARC panelist reports – a summary of the reviewer's opinions and recommendations as well as individual reports from each panelist. SAW/SARC assessment reports are available online at http://www.nefsc.noaa.gov/nefsc/publications/series/crdlist.htm. The CIE review reports and assessment reports can be found at *http://www.nefsc.noaa.gov/nefsc/saw/*".

The 50th SARC was convened in Woods Hole at the Northeast Fisheries Science Center, June 1-5, 2010 to review three assessments: goosefish (also called monkfish; *Lophius americanus*), sea scallop (*Placopecten magellanicus*), and pollock (*Pollachius virens*)). CIE reviews for SARC50 were based on detailed reports produced by NEFSC Assessment Working Groups. This Introduction contains a brief summary of the SARC comments, a list of SARC panelists, the meeting agenda, and a list of attendees (Tables 1 – 3). Maps of the Atlantic coast of the USA and Canada are also provided (Figures 1 - 5).

Outcome of Stock Assessment Review Meeting:

The SARC review committee accepted the monkfish assessment, but expressed serious concerns regarding the high levels of uncertainty throughout the assessment. There is considerable uncertainty in estimates of stock size, recruitment, fishing mortality, biological reference points, stock status determination, and stock projections. There is a large retrospective pattern in the model for the northern management area. It is possible that similar uncertainties exist in the southern management area. Sources of uncertainty in the assessment are neither well characterized nor documented. The scientific basis of the redefined reference points is adequate, but they are uncertain given their dependence upon the uncertain assessment model. Under both the unadjusted and adjusted retrospective scenarios, monkfish in both the northern and southern management areas are not overfished and overfishing is not occurring. The causes of the retrospective patterns in the models need to be determined.

The Panel accepted the sea scallop assessment. The assessment was rigorous and it was well supported by the available information. Strong analytical frameworks were defined for estimating fishing mortality, stock biomass and recruitment (CASA model), for defining biological reference points (SYM model) and for performing stock projections to inform ABC decisions (SAMS model). An innovative approach was developed for quantifying uncertainties around BRPs relative to exploitation levels, facilitating the incorporation of risk assessment into fishery management decisions. The stock is not overfished, and overfishing is not occurring, although the probability of overfishing is only marginally less than 50%. The SAMS model allows complex spatial management scenarios to be addressed. The principal uncertainty in the assessment concerns whether the current high productivity levels will continue in the future.

The Panel accepted the pollock assessment. The new assessment method (ASAP) is a significant improvement over the previous method (AIM). There is significant concern over the presumed large and as of yet unobserved adult biomass (i.e. cryptic biomass) and its implications for fishery management. For the future, the Panel recommends a risk analysis approach to determine the consequences to management of different assumptions about exploitable biomass. The Panel emphasizes the need for field evidence to document whether the cryptic biomass exists. Based on the assessment the stock is not overfished and overfishing is not occurring. This conclusion is robust to the assumptions about the shape of the survey selectivity curve. However, the Biological Reference Points (BRPs) are sensitive to the assumed shape of the selectivity curve, which has consequences for the projection results.

CIE review reports can be found at *http://www.nefsc.noaa.gov/nefsc/saw/* under the heading “SARC 50 Panelist Reports”.

Table 1. 50th Stock Assessment Review Committee Panel.

50th Northeast Regional Stock Assessment Workshop (SAW 50)
Stock Assessment Review Committee (SARC) Meeting

June 1-5, 2010
Woods Hole MA

SARC Chairman (NEFMC SSC):
Mr. Bob O'Boyle (SARC50 Chair)
Beta Scientific Consulting Inc. 1042 Shore Dr.
Bedford, Nova Scotia
Canada B4A 2E5
betasci@eastlink.ca

SARC Panelist (NEFMC SSC):
Dr. Patrick Sullivan
214 Fernow Hall
Cornell University
Ithaca, NY 14853
e.mail: pjs31@cornell.edu

SARC Panelists (CIE):
Dr. Michael Bell
International Centre for Island Technology
Heriot-Watt University
Old Academy
Stromness, Orkney, KW16 3AW, UK
E-mail: m.c.bell@hw.ac.uk

Dr. Kurtis Trzcinski
Population Ecology Division
Bedford Institute of Oceanography
Department of Fisheries and Oceans
P.O. Box 1006
Dartmouth, Nova Scotia
B2Y 4A2, Canada
E-mail: Kurtis.Trzcinski@dfo-mpo.gc.ca

Mr. John Wheeler
Department of Fisheries and Oceans
Science Branch, Aquatic Resource Div.
Northwest Atlantic Fisheries Centre
St. John’s, Newfoundland
E-mail: WheelerJ@DFO-MPO.GC.CA

Table 2. Agenda, 50th Stock Assessment Review Committee Meeting.

50th Northeast Regional Stock Assessment Workshop (SAW 50)
Stock Assessment Review Committee (SARC) Meeting
June 1-5, 2010

Stephen H. Clark Conference Room – Northeast Fisheries Science Center
Woods Hole, Massachusetts

FINAL AGENDA* **(version: 27 May 2010)**

TOPIC	PRESENTER(S)	SARC LEADER	RAPPORTEUR

Tuesday, June 1

8:45-9 AM
Opening
Welcome — **James Weinberg**, SAW Chair
Introduction — **Robert O'Boyle**, SARC Chair
Agenda
Conduct of Meeting

9-11 Assessment Presentation (A. Monkfish)	**Anne Richards**	**John Wheeler**	**M. Traver**
11-11:15 Break			
11:15 -Noon SARC Discussion w/ presenters (A. Monkfish)	**Robert O'Boyle**, SARC Chair		**M. Traver**
Noon – 1:15 Lunch			
1:15–3:30 Assessment Presentation (B. Sea Scallop)	**Dvora Hart** **Larry Jacobson**	**Mike Bell**	**T. Chute**
3:30-3:45 Break			
3:45-5:30 SARC Discussion w/ presenters (B. Sea Scallop)	**Robert O'Boyle**, SARC Chair		**T. Chute**

Wednesday, June 2

8:45-10:45 Assessment Presentation (C. Pollock)	**Liz Brooks**	**Kurtis Trzcinski**	**J. Blaylock**
10:45-11 Break			

11 -Noon SARC Discussion w/ presenters (C. Pollock)
Robert O'Boyle, SARC Chair **J. Blaylock**
Noon – 1:15 Lunch

1:15–3:15 Revisit w/ presenters (A. Monkfish)
Robert O'Boyle, SARC Chair **L. Alade**
3:15-3:30 Break

3:30-5:30 Revisit w/ presenters (B. Sea Scallop)
Robert O'Boyle, SARC Chair **T. Chute**

7:00 (social)

Thursday, June 3

8:45-10:45 Revisit w/ presenters (C. Pollock)
Robert O'Boyle, SARC Chair **J. Nieland**
10:45–11 Break

11-Noon Review/edit Assessment Summary Report (C. Pollock) **J. Nieland**
Robert O'Boyle, SARC Chair

Noon–1:15 Lunch

1:15–3 cont. Review/edit Assessment Summary Report (C. Pollock)
Robert O'Boyle, SARC Chair **J. Nieland**

3–3:15 Break

3:15–5:45 PM Review/edit Assessment Summary Report (A. Monkfish)
Robert O'Boyle, SARC Chair **Alade/Traver**

Friday, June 4

9-11:30 Review/edit Assessment Summary Report (B. Sea Scallop)
Robert O'Boyle, SARC Chair **T. Chute**

11:30–1:00 Lunch

1–5:30 PM SARC Report writing. (closed meeting)

Saturday, June 5

9–5:30 PM SARC Report writing. (closed meeting)

*Times are approximate, and may be changed at the discretion of the SARC chair. The meeting is open to the public, except where noted.

Table 3. 50th SAW/SARC, List of Attendees

Name	Affiliation	email
Andrea Toran	NEFSC	andrea.toran@noaa.gov
Maggie Raymond	AFM	
Vidar Wepestad	MDF	
Michele Traver	NEFSC	michele.traver@noaa.gov
Steve Cadrin	NEFSC	steven.cadrin@noaa.gov
JJ Maguire	MDF	jjmaguire@sympatico.ca
Julie Nieland	NEFSC	julie.nieland@noaa.gov
Ron Smolowitz	FSF	cfarm@capecod.net
Crista Bank	UMD & SMAST	cbank@umassd.edu
Chris Legault	NEFSC	chris.legault@noaa.gov
Jason Link	NEFSC	jason.link@noaa.gov
Paul Nitschke	NEFSC	paul.nitschke@noaa.gov
Michael Palmer	NEFSC	michael.palmer@noaa.gov
Larry Jacobson	NEFSC	larry.jacobson@noaa.gov
Liz Brooks	NEFSC	liz.brooks@noaa.gov
Allison McHale	NERO	allison.mchale@noaa.gov
Kathy Sosebee	NEFSC	katherine.sosebee@noaa.gov
Jessica Blaylock	NEFSC	jessica.blaylock@noaa.gov
Mark Terceiro	NEFSC	mark.terceiro@noaa.gov
Phil Haring	NEFMC	pharing@nefmc.org
Loretta O'Brien	NEFSC	loretta.o'brien@noaa.gov
Susan Wigley	NEFSC	susan.wigley@noaa.gov
Mike Bell	ICIT/Heriot-Watt	m.c.bell@hw.ac.uk
Pat Sullivan	Cornell	pjs31@cornell.edu
Kevin Stolcking	UMD & SMAST	kevin.stolcking@umassd.edu
Saang-Yoon Hyun	UMD & SMAST	shyun@umassd.edu
Cate O'Keefe	UMD & SMAST	cokeefe@umassd.edu
Karen Bolles	HABCAM	kbolles03@yahoo.com
Alan Seaver	NEFSC	alan.seaver@noaa.gov
David Rudders	VIMS	rudders@vims.edu
Richard Taylor	HABCAM	rtaylor@cove.com
Drew Minkiewicz	FSF	aminkiewicz@
Deidre Boelke	NEFMC	dboelbe@nefmc.org
Jess Melgey	NEFMC	jmelgey@nefmc.org
Michael Sissenwine	WHOI/ICES/NEFSC/SS	m_sissenwine@surfglobal.net
Bill DuPaul	VIMS	dupaul@vims.edu
James Weinberg	NEFSC	james.weinberg@noaa.gov
Paul Rago	NEFSC	paul.rago@noaa.gov
Bob O'Boyle	Meeting Chair	betasci@eastlink.ca
Kurtis Trzcinski	DFO	kurtis.trzcinski@dfo-mpo.gc.ca
John Wheeler	DFO	wheelerj@dro-mpo.gc.ca
Anne Richards	NEFSC	anne.richards@noaa.gov
Dvora Hart	NEFSC	dvora.hart@noaa.gov
Toni Chute	NEFSC	toni.chute@noaa.gov
Tom Warren	NMFS	thomas.warren@noaa.gov
Mike Russo		russom447@aol.com
Billy Revillini	NEFSC	wrivellini@colgate.edu
Daniel Goethel	UMD & SMAST	dgoethel@umassd.edu
Anne Hawkins	NEFMC	ahawkins@nefmc.org
Frank Almeida	NEFSC	frank.almeida@noaa.gov
Richard Merrick	NEFSC	richard.merrick@noaa.gov
Jon Deroba	NEFSC	jon.deroba@noaa.gov
Tom Nies	NEFMC	tnies@nefms.org
Larry Alade	NEFSC	larry.alade@noaa.gov
Maurice Crawford	Elizabeth City State U	
Kevin McIntosh	NEFSC	kevin.mcintosh@noaa.gov
Gary Shepherd	NEFSC	gary.shepherd@noaa.gov

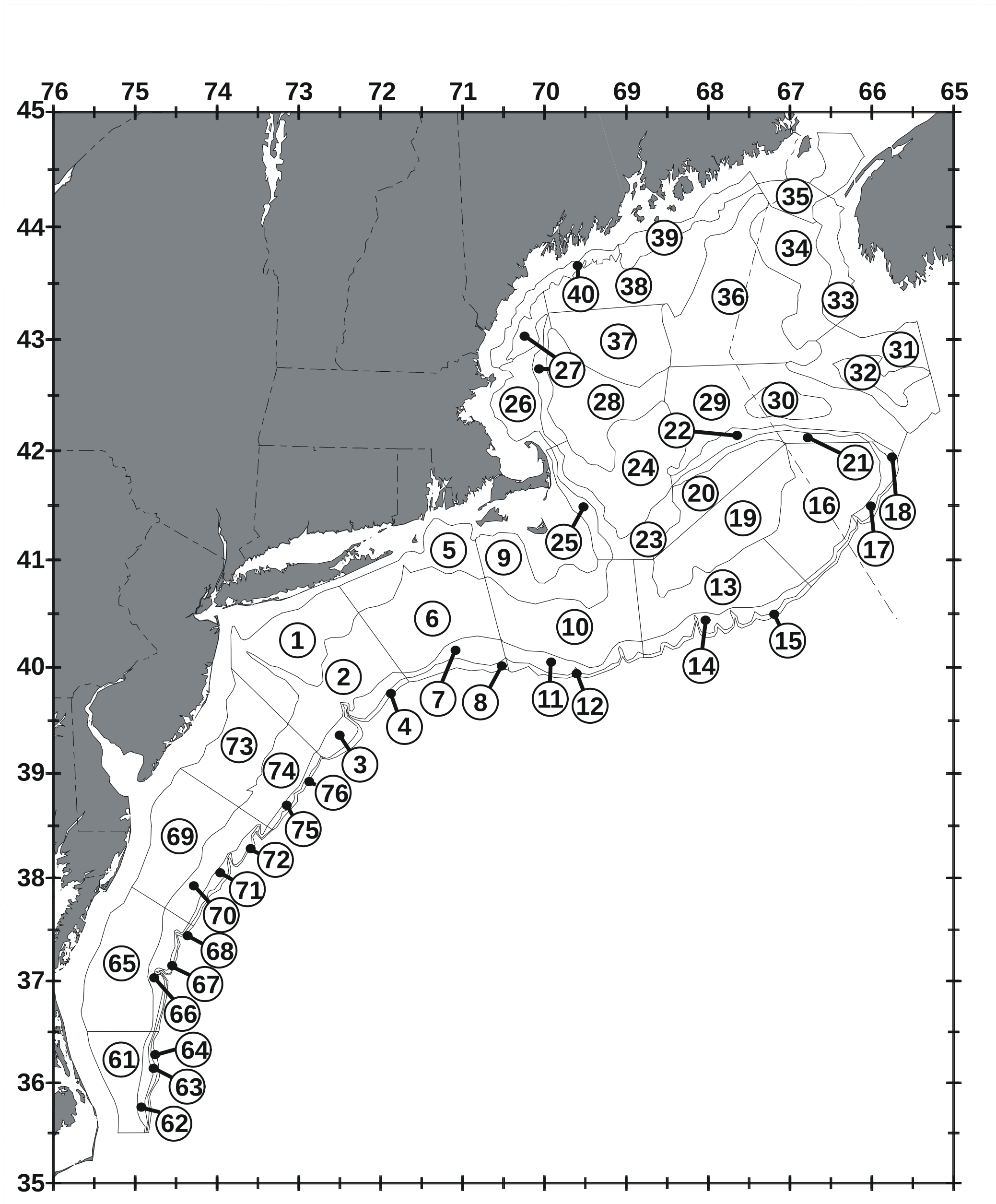

Figure 1. Offshore depth strata sampled during Northeast Fisheries Science Center bottom trawl research surveys.

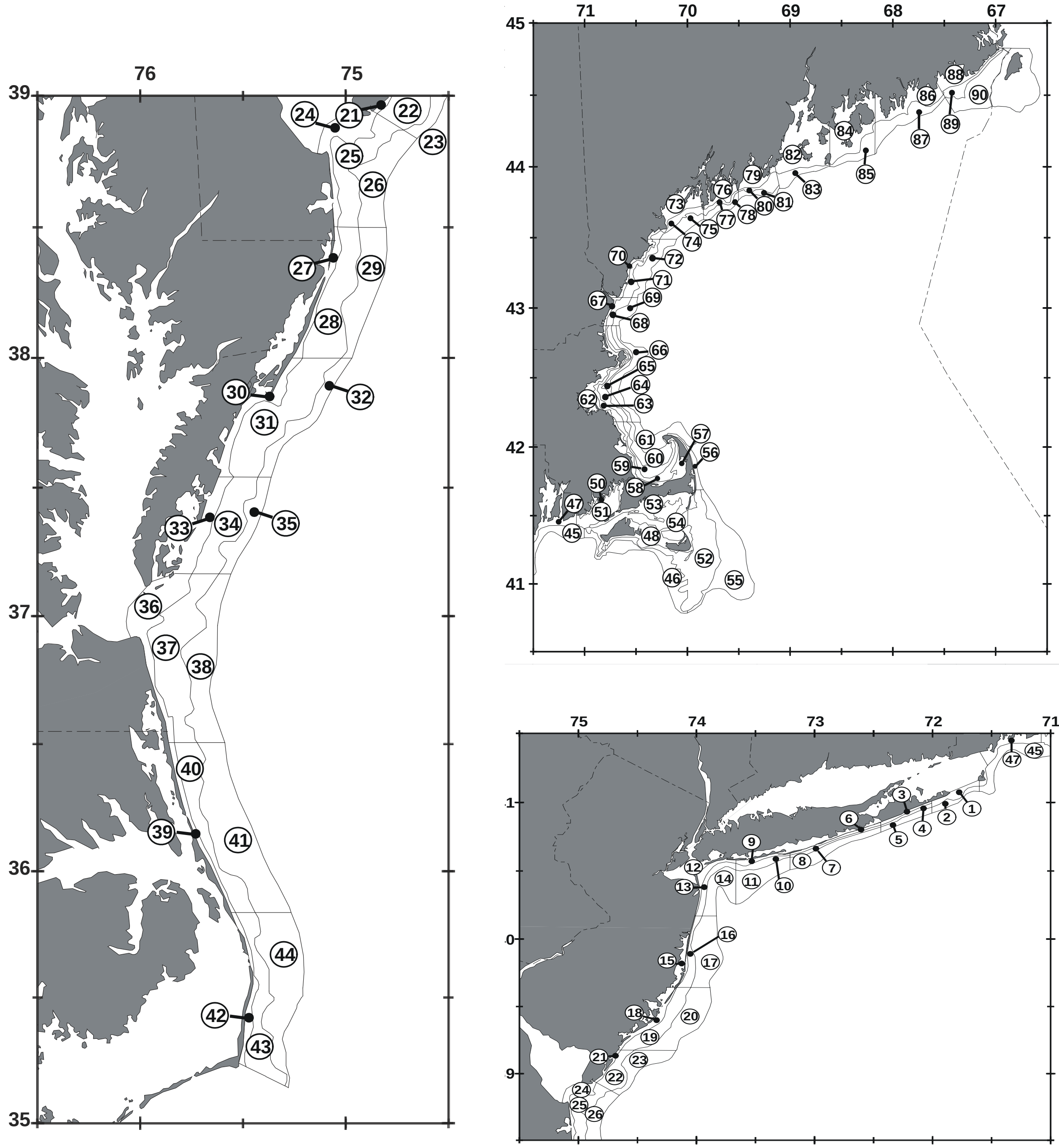

Figure 2. Inshore depth strata sampled during Northeast Fisheries Science Center bottom trawl research surveys.

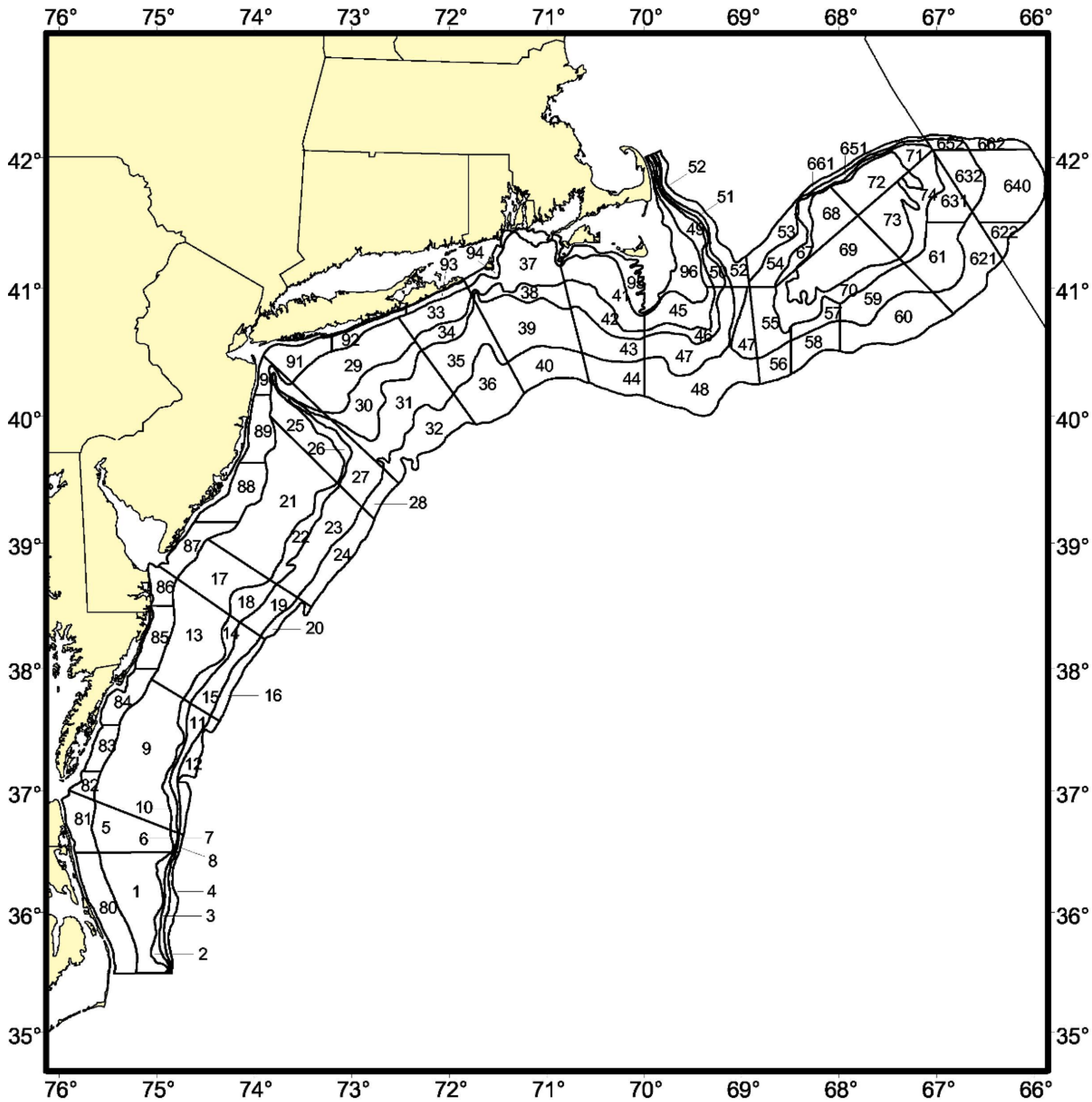

Figure 3. Depth strata sampled during Northeast Fisheries Science Center clam dredge research surveys.

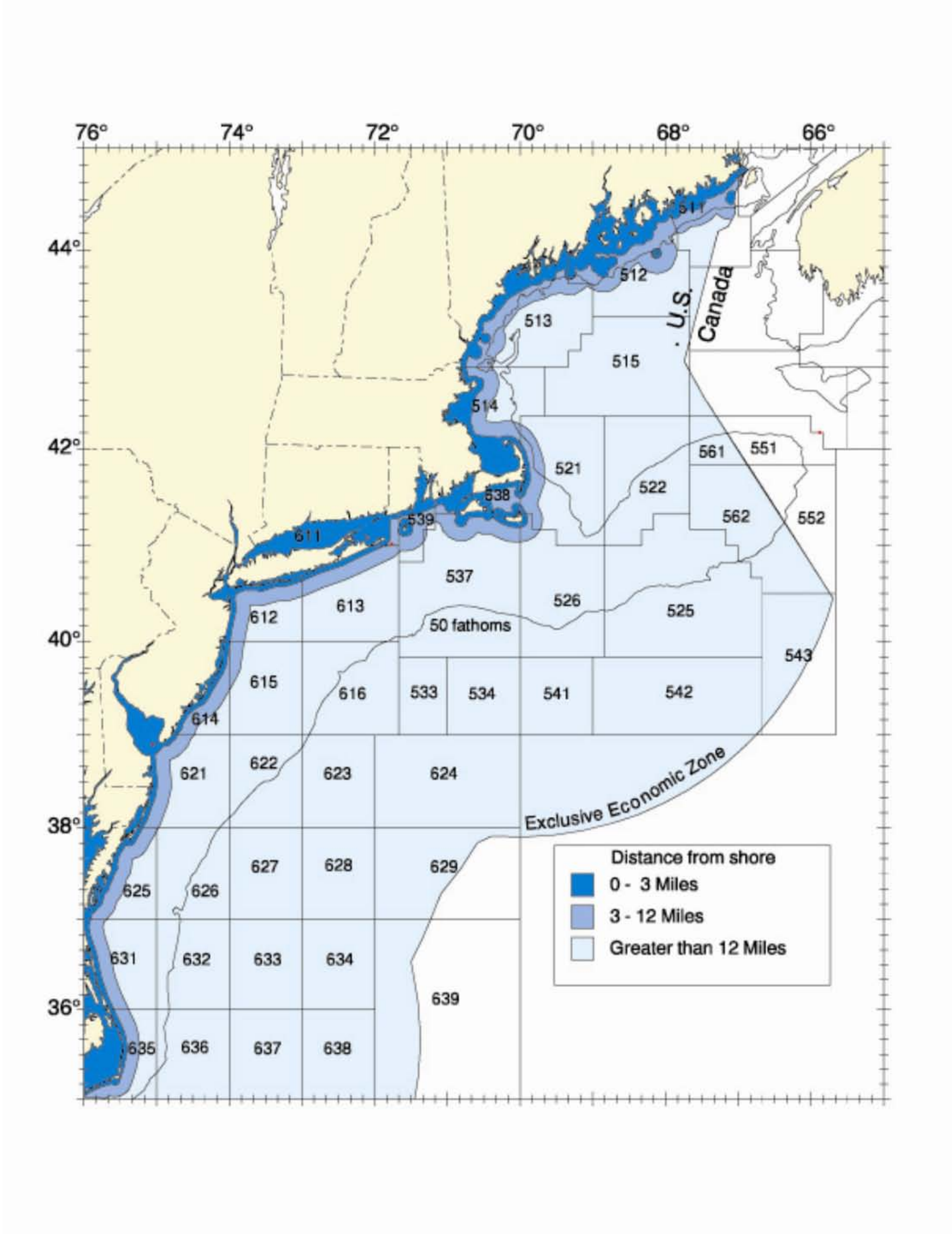

Figure 4. Statistical areas used for reporting commercial catches.

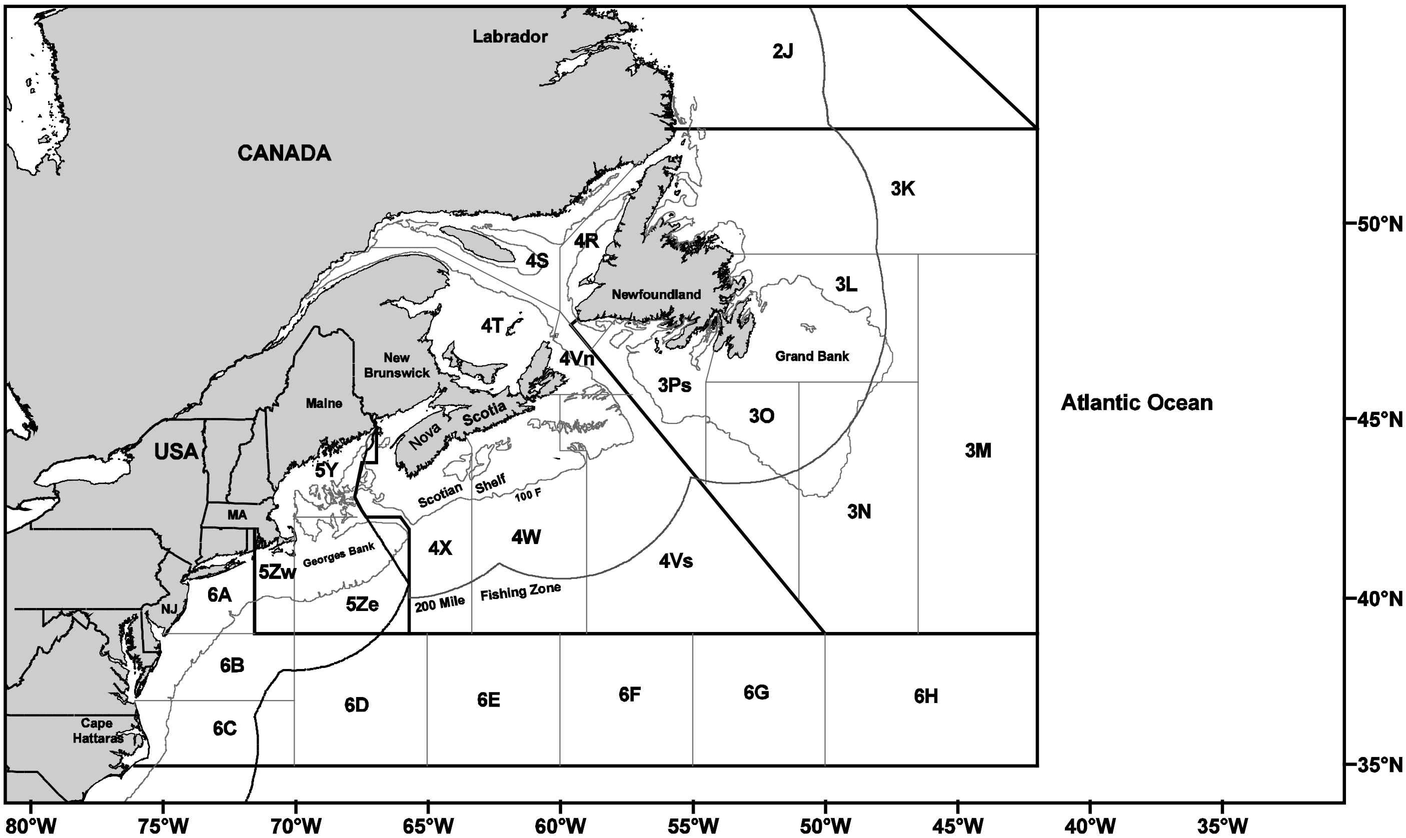

Figure 5. Catch reporting areas of the Northwest Atlantic Fisheries Organization (NAFO) for Subareas 3-6.

A. MONKFISH (GOOSEFISH) STOCK ASSESSMENT FOR 2010

SAW50 Editor's Note: The SAW Chair has added comments to this monkfish assessment report, all of which use bold italicized text. These comments are included to present some opinions and decisions of the SARC50 peer review panel. The comments inserted here do not replace and are not a substitute for the complete set of reviewer reports that are available online from the SAW/SARC website (http://www.nefsc.noaa.gov/nefsc/saw/ in the SAW50 section).

Southern Demersal Working Group (WG)

The Southern Demersal Working Group prepared the stock assessment. The WG met during April 12-15, 2010 at the Northeast Fisheries Science Center, Woods Hole. MA, USA, with the following participants:

Larry Alade	NMFS NEFSC
Crista Bank	UMASS SMAST
Eleanor Bochenek	Rutgers University
Steve Cadrin	NMFS NEFSC/NEFMC SSC; via Webex
Trisha DeGraaf	Maine DNR
Phil Haring	NEFMC
Jason Link	NMFS NEFSC
J -J Maguire	Halieutikos, Inc., Monkfish Defense Fund, NEFMC SSC; via Webex
Allison McHale	NMFS NERO
Paul Nitschke	NMFS NEFSC (SCALE model)
Mike Palmer	NMFS NEFSC
Paul Rago	NEFSC NMFS
Anne Richards	NMFS NEFSC (assessment lead)
Fred Serchuk	NMFS NEFSC
Katherine Sosebee	NMFS NEFSC
Nils Stolpe	Monkfish Defense Fund; via Webex
Sandy Sutherland	NMFS NEFSC
Mark Terceiro	NMFS NEFSC (WG chair)
Michele Traver	NMFS NEFSC
Vidar Wespestad	Monkfish Defense Fund

SARC 50 Monkfish Terms of Reference

1. Characterize the commercial catch including landings, effort, LPUE and discards. Describe the uncertainty in these sources of data.
2. Report results of 2009 cooperative monkfish survey and describe sources of uncertainty in the data and results.
3. Characterize other survey data that are being used in the assessment (e.g., regional indices of abundance, recruitment, length data, state surveys). Describe the uncertainty in these sources of data.
4. Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series, and characterize the uncertainty of those estimates.
5. Update or redefine biological reference points (BRPs; estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, and $F_{MSY;}$ and estimates of their uncertainty). Comment on the scientific adequacy of existing and redefined BRPs.
6. Evaluate stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 5).
7. Evaluate monkfish diet composition data and its implications for population level consumption by monkfish.
8. Develop and apply analytical approaches and data that can be used for conducting single and multi-year stock projections and for computing candidate ABCs (Acceptable Biological Catch; see Appendix to the TORs).
    a. Provide numerical short-term projections (through 2016). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. In carrying out projections, consider a range of assumptions to examine important sources of uncertainty in the assessment.
    b. Comment on which projections seem most realistic, taking into consideration uncertainties in the assessment.
    c. Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC.
9. Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in recent SARC reviewed assessments and review panel reports. Identify new research recommendations.

Executive Summary

The Southern Demersal Working Group (SDWG) met in April 2010 to develop stock assessments for the northern, southern and combined management areas of the U.S. fishery resource. The SDWG met within the process of Northeast SAW 50 and addressed 10 terms of reference, as follows.

1. Characterize the commercial catch including landings, effort, LPUE and discards. Describe the uncertainty in these sources of data.

Reported total landings (live weight) increased from an average of 2,500 mt in the 1970s to 8,700 mt in the 1980s, 23,000 mt in the 1990s, 22,000 mt from 2000-2005 and 11,600 mt during 2006-2009. Total landings have declined since 2003 due to management regulations including TACs during 2007-2009 of 5,000 mt in the northern area and 5,100 mt in the southern area. Landings in 2009 were 3,255 mt in the northern area and 5,302 mt in the southern area.

Estimated total discards of monkfish during 1989-2009 have ranged between 1,600 mt (1992) and 7,500 mt (2001) per year, with a long-term discard/kept ratio of 0.15 (northern and southern areas combined). Discard rates have been highest in the scallop dredge fisheries in the southern area, and lowest in gillnets in both areas. Discard ratios and discard levels (mt) increased in both areas after 2000, and have since declined somewhat (overall discard/kept ratio for 2000-2004 =0.20; for 2005-2009=0.17).

Length composition of landings was fairly stable during 2002-2009, with modal lengths ~52 cm in the north, ~65 cm in the south and few fish larger than 85 cm in either area. Recent decreases in landings have not resulted in a broadening of the size composition of landings.

Evaluating trends in effort or catch rates in the monkfish fishery is difficult because much of the catch is taken in multi-species fisheries, and defining targeted monkfish trips is problematic. Furthermore, programmatic changes from port interviews (1980-1993) and logbooks (1994-2006) make temporal comparison of effort statistics difficult. CPUE estimated from observed tows has declined in the north since 2003-2005 and remained stable or declined since 2004 in the south; however estimates of CPUE have a high variance and may not be reliable.

Estimation of total catch for monkfish has several sources of uncertainty. Before 1980, fishery removals were primarily bycatch, but most were unreported. Therefore, evaluation of fishery development is difficult, leading to problems interpreting the state of the resource in the early years of the marketed fishery. Since 1980, the quality of landings estimates generally increased, but the series includes under-reporting and difficulties converting landed products to live weight. Historical under-reporting of landings should be considered in the interpretation of this series.

There is no information on the magnitude of discards prior to 1989. The SDWG assumed that discard rates before 1989 were similar to discard: kept ratios observed in later years; this may be problematic if discard rates were lower in later years because markets had developed. The quality of discard data generally increased in the 1989-2009 observer time series, as a result of increasingly greater coverage of fleets and improved protocols, but there were some unsampled portions of the fishery (e.g., some half-year periods in which entire gear-types were not sampled).

Characterizing size and age composition of the catch also has considerable sources of uncertainty. Length sampling by fishery observers started earlier in the time series than sampling of landings in ports (1989 vs. 1996) and was more comprehensive (NEFSC 2007a); however, sampling intensity in most years is adequate only for estimation on a half-year basis. Age samples from at-sea observers have not been processed and are on hold until the ageing method is validated.

2. Report results of 2009 cooperative monkfish survey and describe sources of uncertainty in the data and results.

A cooperative monkfish survey was conducted during Feb-Apr 2009 using two industry trawlers and 3 nets (2 flat, 1 rockhopper). The survey design differed slightly from previous cooperative surveys (in 2001, 2004) because sampling effort was allocated in proportion to stratum area rather than to spatial patterns of fishing effort. The estimates of area swept population size and biomass for 2009 are lower than those estimated from earlier cooperative monkfish surveys (2001, 2004). The estimated population length composition was similar among cooperative surveys with a mode around 34 cm in the NMA and a bimodal distribution (~32 cm and ~52 cm) in the SMA. Length frequency composition data from the 2009 cooperative survey were input into the final SCALE assessment model. Major sources of uncertainty include timing of the survey with respect to spring onshore migrations and accuracy of net efficiency estimates from depletion experiments.

3. Characterize other survey data that are being used in the assessment (e.g., regional indices of abundance, recruitment, length data, state surveys). Describe the uncertainty in these sources of data.

Several surveys sample monkfish and provide time series of relative abundance. However, no single survey (with the exception of the new NEFSC survey on the FSV Bigelow) catches large numbers of monkfish throughout either management area. The NEFSC spring and autumn bottom trawl surveys provide long-term series that sample the entire continental shelf to 300m depth, but they only catch approximately 100 monkfish in each management area per year. The NEFSC winter bottom trawl survey and scallop survey, the ASMFC shrimp survey, and the ME/NH inshore survey catch considerably more monkfish, but are shorter series, and sample only a portion of either management area.

Within the northern management area, broad trends in stock size are consistent among the five surveys conducted there. Biomass fluctuated without trend from 1963 to the early 1980s, but declined thereafter to near historic lows during the 1990's when landings reached their peak. Biomass indices increased from 2000 to 2004, but have generally decreased since then. Abundance indices in the north fluctuated without trend during 1963-1998 but spiked during 2000-2002, reflecting a strong 1999 year class.

General trends in stock size in the southern area are also consistent among surveys. Survey biomass and abundance indices were high during the mid-1960s, fluctuated around an intermediate level during the 1970s and mid-1980s, then declined to low levels since the late 1980s. Biomass indices increased slightly around 2002 but have returned to lower levels since then.

Size-based indices of abundance indicate relatively strong recruitment in the northern area during the 1990s and variable but stable recruitment in the south. Length distributions gradually truncated from the 1960s to1990, and the median size of monkfish in survey catches has remained fairly constant since the early 1990s.

4. Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series, and characterize the uncertainty of those estimates.

Fishing mortality rates, recruitment and stock sizes were estimated using the SCALE statistical catch-at-length model. Estimated F in 2009 was 0.10 in the north and 0.07 in the south (0.05 combined areas). Estimated total biomass in 2009 was 66,062 mt in the north and 131,218 mt in the south (255,326 mt, combined areas). In the north, the strongest year classes were produced in 1997-1999; recruitment was generally below average in the 1980s, and has been about average since 2001. In the south, the strongest year classes were produced in 1992, 1997, and 2002; recruitment has been below average since 2004. Based on the combined-areas model, the strongest year classes were produced in 1997-1999 and recruitment has been below average since 2004.

Uncertainty in the estimates of stock size, recruitment and F stems from poorly known input data, including under-reported landings and unknown discards during the 1980s, and incomplete understanding of key biological parameters such as age and growth, longevity, natural mortality, sex ratios and stock structure, and the relatively short reference time frame (1980-2006) of the model. Further, the population models for all areas exhibit retrospective patterns that are strongest for the 2002-2006 terminal years and weaker for the 2007-2008 terminal years. The retrospective patterns are strongest for the northern area, weakest for the southern area, and intermediate for the model of combined areas.

SAW50 Editor's note: In view of the short time available for the review, the SARC50 panel declined to review the combined-areas model as it addressed a Research Recommendation rather than a Term of Reference, and because management is based on the two-areas model.

The SARC50 panel acknowledged the high degree of uncertainty in estimates from the SCALE model due to data limitations, poorly understood monkfish biology (growth, natural mortality, stock structure), and the strong retrospective pattern in the northern area.

5. Update or redefine biological reference points (BRPs; estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, and $F_{MSY;}$ and estimates of their uncertainty). Comment on the scientific adequacy of existing and redefined BRPs.

The 2007 NEFSC assessment recommended new reference points based on a revised yield-per-recruit analysis (using M=0.3) and on the results of the SCALE length-tuned model that incorporated multiple survey indices and catch data. The new reference biomass levels were based on long term trends in biomass from the SCALE model, and were adopted in Framework 5 (April 2008). The current assessment updates the SCALE model and estimates new reference points based on the methods adopted in NEFSC (2007a) and using the method applied in the New England groundfish stock complex based on projections of B_{max} at F_{max}. The BRPs all use output from the SCALE model, which is subject to high levels of uncertainty as discussed under TOR 4, therefore the BRPs are also highly uncertain.

The following table summarizes the estimates for each management area and combined areas. Adjusted refers to estimates adjusted for retrospective patterns.

Management	Biomass BRPs in metric tons			
Areas				
North	**BRP**	**Basis**	**DPSWG 2007**	**SDWG 2010**
	Fmax	YPR	0.31	0.43
	Bthreshold	Bloss 1980-2006	65,200	
	Bthreshold	Bloss 1980-2009		41,238
	Bthreshold	0.5*Bmax Projected		26,465
	Bthreshold	0.5*Bmax Proj Adjust		20,643
	Btarget	Bavg 1980-2006	92,200	62,371
	Btarget	Bavg 1980-2009		61,991
	Btarget	Bmax Projected		52,930
	Btarget	Bmax Proj Adjust		41,286
	MSY	Fmax Projected		10,745
South	**BRP**	**Basis**	**DPSWG 2007**	**SDWG 2010**
	Fmax	YPR	0.40	0.46
	Bthreshold	Bloss 1980-2006	96,400	
	Bthreshold	Bloss 1980-2009		99,181
	Bthreshold	0.5*Bmax Projected		37,245
	Bthreshold	0.5*Bmax Proj Adjust		28,461
	Btarget	Bavg 1980-2006	122,500	120,292
	Btarget	Bavg 1980-2009		121,313
	Btarget	Bmax Projected		74,490
	Btarget	Bmax Proj Adjust		56,922
	MSY	Fmax Projected		15,279
Combined	**BRP**	**Basis**	**DPSWG 2007**	**SDWG 2010**
	Fmax	YPR		0.37
	Bthreshold	Bloss 1980-2009		159,715
	Bthreshold	0.5*Bmax Projected		64,501
	Bthreshold	0.5*Bmax Proj Adjust		49,021
	Btarget	Bavg 1980-2009		208,190
	Btarget	Bmax Projected		129,002
	Btarget	Bmax Proj Adjust		98,041
	MSY	Fmax Projected		25,943

SAW50 Editor's note: The SARC50 panel recommended adoption of the biomass reference points based on "Bmax projected" for each management area. The word "adjust" in the table above refers to results that were adjusted for the retrospective pattern. Although the SARC50 panel did not recommend using the "adjusted" values directly, the panel was well aware and very concerned about the lack of model fit.

6. Evaluate stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 5).

Estimates of total biomass for 2006 in both management areas (see table below) were greater than their respective biomass targets, therefore, based on those somewhat uncertain analyses, monkfish in both management areas were not overfished and overfishing was not occurring.

Estimates of total biomass for 2009 in both management areas and the combined area (see table below), were above $B_{threshold}$ and B_{target}, but with a smaller margin in the north than estimated in 2006. These estimates are subject to the same uncertainty as the assessment in 2006.

	Stock Biomass			F					
	North	South	N+S	North	South	N+S	Overfished	Overfishing	Bthreshold Basis
SCALE 2006	119,000	135,000	-	0.09	0.12		no	no	Bloss (1980-2006)
SCALE 2009	66,062	131,218	255,326	0.10	0.07	0.05	no	no	Bloss (1980-2009)

SAW50 Editor's note: The SARC50 panel acknowledged the high degree of uncertainty in estimates from the SCALE model due to data limitations, poorly understood monkfish biology (growth, natural mortality, stock structure), and the strong retrospective pattern in the northern area. This uncertainty affects not only the current estimates of biomass but the estimates of the BRPs as well.

7. Evaluate monkfish diet composition data and its implications for population level consumption by monkfish.

Diet composition, per capita consumption, total consumption, and the amount of prey removed by monkfish were calculated from basic monkfish food habits data. Based on recent energy budgets, the amount of food consumed by monkfish is 0.005-0.02% of all energy flows in the system, and monkfish account for 2-6% of the total consumption by all finfish in the ecosystem (1-4 % in the northern area, 2-8% in the southern area).

The total amount consumed and per capita consumption peaked in the early 1980s for both stocks, driven by larger fish. Monkfish consumption of mackerel and herring is potentially 20-50% of landings, about equal to landings for squids, and potentially greater than the landings of silver hake and skates. Monkfish is an important piscivore in the ecosystem.

8. Develop and apply analytical approaches and data that can be used for conducting single and multi-year stock projections and for computing candidate ABCs (Acceptable Biological Catch; see Appendix to the TORs).
   a. Provide numerical short-term projections (through 2016). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. In carrying out projections, consider a range of assumptions to examine important sources of uncertainty in the assessment.
   b. Comment on which projections seem most realistic, taking into consideration uncertainties in the assessment.

c. Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC.

SCALE model results and AGEPRO projections were used to evaluate stock trends during 2011-2016 with $F=F_{threshold}$ and at proposed ACTs and ABCs assuming stochastic long-term recruitment. The projections indicate that the northern area is the most vulnerable to overfishing or becoming overfished during 2011-2016 if total catches approach the proposed ABC, while the southern area is the least vulnerable.

Projections for the northern area (NMA) are the most likely to be unrealistic, given the uncertainty of stock status due mainly to the relatively strong retrospective observed since 2002. The southern area (SMA) projections are the most likely to be realistic, given the moderate retrospective observed for that area. The combined area projections are intermediate with respect to the current management areas, as the relative scaling of the two populations is maintained when the areas are combined in one model.

SAW50 Editor's note: The SARC50 panel acknowledged the high degree of uncertainty in the projections due to uncertainty in the starting conditions (output from the SCALE model).

9. Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in recent SARC reviewed assessments and review panel reports. Identify new research recommendations.

A list of 26 research recommendations generated since SAW 34 in 2001 was reviewed and results summarized where available. Of these, 14 had either been addressed or were considered no longer relevant. One new recommendation was added by the SDWG in 2010.

Introduction

Life History

Monkfish (*Lophius americanus*), also called goosefish, are distributed in the Northwest Atlantic from the Grand Banks and northern Gulf of St. Lawrence south to Cape Hatteras, North Carolina (Collette and Klein-Macphee 2002). Monkfish may be found from inshore areas to depths of at least 900 m (500 fathoms). Seasonal onshore-offshore migrations occur and appear to be related to spawning and possibly food availability (Collette and Klein-MacPhee 2002).

Monkfish rest partially buried on soft bottom substrates and attract prey using a modified first dorsal fin ray that resembles a fishing pole and lure. Monkfish are piscivorous and commonly eat prey as large as themselves. Despite the behavior of monkfish as a demersal 'sit-and-wait' predator, recent information from electronic tagging suggests seasonal off-bottom movements (Rountree et al. 2006). Growth is rapid at about 10 cm per year, and is similar for both sexes up to age 6 and lengths of around 60 cm (Richards et al. 2008). Few males are found older than age 7, but females can live to 12-14 years or older. Monkfish as large as 138 cm have been captured in NEFSC bottom trawl surveys.

Female monkfish begin to mature at age 4 and 50% of females are mature by age 4.7 (about 41 cm). Males mature at slightly younger ages and smaller sizes (50% maturity at age 4.2 or 37 cm (NEFSC 2002; Richards et al. 2008). Spawning takes place from spring through early autumn, progressing from south to north, with most spawning occurring during the spring and early summer. Females lay a buoyant mucoid egg raft or veil which can be as large as 12 m long and 1.5 m wide and only a few mm thick. The eggs are arranged in a single layer in the veil, and the larvae hatch after about 1-3 weeks, depending on water temperature. The larvae and juveniles spend several months in a pelagic phase before settling to a benthic existence at a size of about 8 cm (Collette and Klein-MacPhee 2002).

Stock Identification

The Fishery Management Plan defines two management areas for monkfish (northern and southern), divided roughly by a line bisecting Georges Bank (Figure A1). The two assessment and management areas for monkfish were defined based on differences in temporal patterns of recruitment (estimated from NEFSC surveys), perceived differences in growth patterns, and differences in the contribution of fishing gear types (mainly trawl, gill net, and dredge) to the landings.

Genetic studies suggest a homogeneous population of monkfish off the U.S. east coast (Chikarmane et al. 2000). Monkfish larvae are distributed over deep (< 300 m) offshore waters of the Mid-Atlantic Bight in March-April, and across the continental shelf (30 to 90 m) later in the year, but relatively few larvae have been sampled in the northern management area (Steimle et al. 1999). NEFSC surveys continue to indicate different recruitment patterns in the two management units in recent years.

The perceived differences in growth were based on studies about 10 years apart and under different stock conditions (Armstrong et al. 1992: Georges Bank to Mid-Atlantic Bight, 1982-1985; Hartley 1995: Gulf of Maine, 1992-1993). Age, growth, and maturity information from the NEFSC surveys and the 2001, 2004 and 2009 cooperative monkfish surveys indicated only minor differences in age, growth, and maturity between the areas (Richards et al., 2008; Johnson et al., 2008). The recent biological evidence (growth, maturity, and genetic information) suggests that use of a single stock hypothesis in the assessment might be appropriate. However, substantial differences in the fisheries exist, and current management maintains separate regulatory areas to accommodate these differences.

The southern deepwater extent of the range of American monkfish (*L. americanus*) overlaps with the northern extent of the range of blackfin monkfish (*L. gastrophysus*; Caruso 1983). These two species are morphologically similar, which may create a problem in identification of survey catches and landings from the southern extent of the range of monkfish. The potential for a problem however is believed to be small. The NEFSC closely examined winter and spring 2000 survey catches for the presence of blackfin monkfish and found none. The cooperative monkfish survey conducted in 2001 caught only eight blackfin monkfish of a total of 6,364 monkfish captured in the southern management area.

Fisheries Management

Commercial fisheries for monkfish occur year-round using gillnets, trawls and scallop dredges. No significant recreational fishery exists. The primary monkfish products are tails, livers and whole gutted fish. Peak fishing activity occurs during November through June, and value of the catch is highest in the fall due to the high quality of livers during this season.

U.S. fisheries for monkfish are managed in the Exclusive Economic Zone (EEZ) through a joint New England Fishery Management Council - Mid-Atlantic Fishery Management Council Monkfish Fishery Management Plan (FMP). The primary goals of the Monkfish FMP are to end and prevent overfishing and to optimize yield and economic benefits to various fishing sectors involved with the monkfish fisheries (NEFMC and MAFMC 1998; Haring and Maguire 2008).

Current regulatory measures vary with type of permit but include limited access, limitations on days at sea, mesh size restrictions, trip limits, minimum size limits and other measures (Tables A1 and A2).

Biological reference points for monkfish were established in the original Fishery Management Plan (FMP), but were revised according to the conclusions of SAW 34 (NEFSC 2002) and again by the Data Poor Stocks Working Group (DPSWG) in 2007 (NEFSC 2007a). The overfishing definition is F_{max}. Prior to 2007, $B_{threshold}$ was defined as one-half of the median of the 1965-1981 3-year average NEFSC autumn trawl survey catch (kg) per tow). After acceptance of an analytical assessment in 2007 (NEFSC 2007a), B_{target} was redefined as the average of total biomass for the model time period (1980-2006) and $B_{threshold}$ as the lowest observed value in the total biomass time series from which the stock has then increased (termed "B_{Loss}"). According to the earlier (survey index-based) reference points, monkfish were overfished and overfishing status could not be determined (NEFSC 2005); however, with adoption of the analytical assessment in 2007, monkfish status was no longer overfished and overfishing was not occurring.

2007 DPSWG Assessment

The DPSWG accepted a length-tuned analytical model (SCALE) for monkfish assessment and status determination, and adopted a value of M=0.3 (vs. M=0.2). However, the WG emphasized that the assessment was highly uncertain due to under-reported landings, unknown discards during the 1980s, incomplete understanding of key biological parameters such as age and growth, longevity, natural mortality and stock structure, the shorter reference time frame (1980-2006) than in previous assessments (1963-2006), and the relatively recent development of the assessment model. The WG concluded that uncertainties in historical catch data precluded application of long-term models that rely on episodes of depletion and recovery to estimate stock size.

2010 SAW 50 Assessment

The 2010 Southern Demersal Working Group (SDWG) updated the SCALE model to assess the status of monkfish using data through 2009. Further developments included examination of retrospective patterns in the SCALE estimates, and development of short-term stochastic age-based projections. Data from a cooperative monkfish survey conducted during winter/spring of 2009 were analyzed and included in the assessment model, along with data collected on the new NEFSC survey vessel, starting in spring 2009, which was adjusted using calibration coefficients developed for monkfish. Length frequency composition data from the 2009 cooperative survey were input into the final SCALE assessment model.

SAW50 Editor's note: The SARC50 panel discussed the relative merits of adjusting for retrospective patterns and decided against making a direct adjustment for the pattern in the current assessment.

TOR 1. Characterize the Commercial Catch including landings, effort, LPUE and discards. Describe the uncertainty in these sources of data.

Landings

Landings statistics for monkfish are sensitive to conversion from landed weight to live weight, because a substantial fraction of the landings occur as tails only (or other parts). The conversion of landed weight of tails to live weight of monkfish in the NEFSC weigh-out database is made by multiplying landed tail weight by a factor of 3.32. Recently concerns have been raised that monkfish landings reported as ‘round’ (no conversion) may actually be ’head-on, gutted’, which has a conversion factor of 1.14, in which case live weight of landings would be underestimated. Assuming all landings classified as ‘round’ are actually ‘head-on, gutted’, the difference in live weight landings would be less than 0.8% on average since the ‘round’ category appeared in 1989. The working group concluded that this was not likely an important source of error in the assessment.

Early catch statistics are uncertain, because many of the monkfish caught were sold outside of the dealer system or used for personal consumption until the mid-1970s. For 1964 through 1989, there are two potential sources of landings information for monkfish; the NEFSC ‘weigh-out’ database, which consists of fish dealer reports of landings, and the ‘general canvass’ database, which contains landings data collected by NMFS port agents (for ports not included in the weigh-out system) or reported by states not included in the weigh-out system (Table A3). All landings of monkfish are reported in the general canvass data as ‘unclassified tails.’ Consequently, some landed weight attributable to livers or whole fish in the canvass data may be inappropriately converted to live weight. This is not an issue for 1964-1981 when only tails were recorded in both databases. For 1982-1989, the weigh-out database contains market category information which allows for improved conversions from landed to live weight. The two data sources produce the same trends in landings, with general canvass landings slightly greater than weigh-out landings. It is not known which of the two measures more accurately reflects landings, but the additional data sources suggest that the general canvass is most reliable for 1964-1981 landings, whereas the availability of market category details suggest that the weigh-out database is most reliable for 1982-1989.

Beginning in 1990, most of the extra sources of landings in the general canvass database were incorporated into the NEFSC weigh-out database. However, North Carolina reported landings of monkfish to the Southeast Fisheries Science Center and until 1997 these landings were not added to the NEFSC general canvass database. Since these landings most likely come from the southern management area, they have been added to the weigh-out data for the southern management area for 1977-1997 for the landings statistics used for stock assessment.

Beginning in July 1994, the NEFSC commercial landings data collection system was redesigned to consist of vessel trip reports (VTR) and dealer weigh-out records. The VTRs include area fished for each trip which is used to apportion dealer-reported landings to statistical areas. The northern management area includes statistical areas 511-515, 521-523 and 561; and the southern management area includes areas 525-526, 562, 537-543 and 611-636 (Figure A1). Each VTR trip should have a direct match in the dealer data base, but this is not always true. VTR records with no matching dealer landings were excluded, but dealer landings with no matching VTR were included in landings statistics, apportioning the unmatched landings to management area using proportions calculated from matched trips pooled over gear, state and quarter.

Total U.S. landings (live weight) remained at low levels until the middle 1970s, increasing less than 1,000 mt to around 6,000 mt in 1978 (Table A3, Figure A2). Annual landings remained stable at between 8,000 and 10,000 mt until the late 1980s. Landings increased from the late 1980s to over 20,000 mt per year 1992-2004, peaking at 28,500 mt in 1997. Landings have declined steadily since 2003, to 8,600 mt in 2009. By region, landings began to increase in the north in the mid-1970s, and began to increase in the south in the late

1970s. Most of the increase in landings during the late 1980s through mid-1990s was from the southern area. Historical under-reporting of landings shold be considered in the interpretation of this series.

Trawls, scallop dredges and gill nets are the primary gear types that land monkfish (Table A4, Figure A3). Trawls have contributed approximately half of the landings. Prior to 1994, gillnets contributed less than 10% of total landings, but landings from gillnets generally increased to account for >35% of the recent fishery, with an associated decrease in monkfish landings from the scallop dredge fishery.

Until the late 1990s, total landings were dominated by landings of monkfish tails. From 1964 to 1980 landings of tails rose from 19mt to 2,302mt, and peaked at 7,191mt in 1997 (Table A5). Landings of tails declined after 1997, but are still an important component of the landings. Landings of gutted whole fish have increased steadily since the early 1990s and are now the largest market category on a landed-weight basis. On a regional basis, more tails were landed from the northern area than the southern area prior to the late 1970s (Tables A6 and A7). From 1979 to 1989, landings of tails were about equal from both areas. In the 1990's, landings of tails from the south predominated, but since 2000, landings of tails have been greater in the north.

Beginning in 1982, several market categories were added to the system (Table A5). Tails were broken down into large (> 2.0 lbs), small (0.5 to 2.0 lbs), and unclassified categories and the liver market category was added. In 1989, unclassified round fish were added, in 1991 peewee tails (<0.5 lbs) and cheeks, in 1992 belly flaps, and in 1993 whole gutted fish were added. Monkfish livers have become a very valuable product. Landings of livers increased from 10mt in 1982 to an average of over 600mt during 1998 - 2000. During 1982-1994, ex-vessel prices for livers rose from an average of $0.97/lb to over $5.00/lb, with seasonal variations as high as $19.00/lb. Landings of unclassified round (whole) or gutted whole fish jumped in 1994 to 2,045mt and 1,454mt, respectively; landings of gutted fish continued to increase through 2003. The tonnage of peewee tails landed increased through 1995 to 364mt and then declined to 153mt in 1999 and 4mt in 2000 when the category was essentially eliminated by regulations.

Foreign Landings

Landings (live wt) from NAFO areas 5 and 6 by countries other than the US are shown in Table A3 and Figure A2. Reported landings were high but variable in the 1960s and 1970s with a peak in 1973 of 6,818mt. Landings were low but variable in the 1980s, declined in the early 1990s, and have generally been below 300mt in recent years.

Discard Estimates

Catch data from the fishery observer and VTR databases were used to investigate discarding frequencies and rates. The number of trips with monkfish discards available for analysis varied widely among management areas and gear types (Table A8). In the previous assessment (NEFSC 2007a), three methods were considered for the estimation of discards: 1) observed discard-per-kept-monkfish expanded to total discards using total monkfish landings; 2) observed discard-per-all-kept-catch expanded to total discards using total landings (Rago et al. 2005, Wigley et al. 2007); and 3) observed discard-per-days-absent expanded to total discards using total days-absent (Rago et al. 2005, Wigley et al. 2007). All three methods were done on a gear, half-year and management area basis. The effort-based method (#3) was considered inappropriate, because much of the monkfish is bycatch taken incidentally or targeted on a tow-by-tow basis rather than on a trip basis. Predicting discards using kept catch assumes a linear relationship between kept and discarded catch and no discarding when there is no catch (i.e., the linear relationship passes through the origin). Inspection of the relationship between observed monkfish discards and monkfish kept (method #1) and total catch (method #2) by gear and year indicated weak correlation in general, but the relationships between kept and discarded monkfish (method #1) for trawls and gillnets conformed to the statistical assumptions best (NEFSC 2007a). Therefore, discard estimates were based on discard-to-kept-monkfish for trawls and gillnets but were based on discard-per-all-kept-catch for shrimp trawls and dredges, which do not currently target monkfish. This method, (NEFSC 2007a) was continued in the current assessment.

Discards for 1980-1988 (before observer sampling) were estimated by applying average discard ratios by management area and gear type (trawl, shrimp trawl, gillnet, dredge) from 1989-1991 to landings for 1980-1988. If insufficient samples were available, additional years of observer data were included until a sample size (number of trips) of at least 20 was reached. The resulting time periods entering the 1980-1988 discard ratio estimates were as follows:

Area	**Shrimp Trawls**	**Trawls**	**Gillnets**	**Dredges**
North				
Years included	1989-1991	1989-1991	1989-1991	1992-1997
Number of trips	124	180	852	20
South				
Years included	n/a	1989-1991	1991-1992	1991-1993
Number of trips		231	103	30

The overall annual discard ratio (discarded monk / kept monk) decreased in the northern area, from an average of 16% of total catch in the 1980s to an annual average of 8% during 2002-2006, but was slightly higher on average (~10%) during 2007-2009 (Table A9, Figure A4). The proportion of discards in the southern area generally increased since 1980, with an annual average of 23% during 2002-2006, but a slight decrease during 2007-2009 (to ~14%) (Table A9, Figure A5). Gill nets consistently have had the lowest discard ratios. Some of the trends in discarding may reflect imposition of size limits starting in 2000 and decreased trip limits in the south starting in 2002. The DPSWG (NEFSC 2007a) noted a potential bias in discard estimates due to increased observer sampling in the multispecies groundfish fishery. Monkfish discard rates may differ between the directed monkfish fisheries and bycatch fisheries. The most frequent discard reasons were that fish were too small for regulations or the market. The estimates of total catch for 1980-2009 are shown in Figure A6 and Table A10.

Size and Age Composition of U.S. Catch

Tail lengths were converted to total lengths using relations developed by Almeida et. al.(1995). As in NEFSC (2007a), length composition of landings and discard were estimated from fishery observer samples by management area, year, gear-type (trawls, dredges and gillnets) and catch disposition (kept or discarded; Figures A7 – A13). Observer sampling data for December 2009 were not yet available, so the sample set for 2009 is incomplete. Landings in unknown gear categories were allocated proportionately to the 3 major gear types before assigning lengths. The stratification used for assigning lengths within area and gear type for

2007-2009 is shown in Table A11. Discards were generally between 20-40 cm, while kept fish were greater than 40 cm; however, there were some exceptions to this pattern in recent years.

Age composition of the catch was not estimated for 2007-2009 due to uncertainties in the aging method that were highlighted during the previous assessment (NEFSC 2007a) and because the operational model for monkfish (SCALE) is length-based.

Effort and CPUE

Evaluating trends in effort or catch rates in the monkfish fishery is difficult for several reasons. Much of the catch is taken in multi-species fisheries, and defining targeted monkfish trips is difficult. There have been programmatic changes in data collection from port interviews (1980-1993) to logbooks (1994-2009), and comparison of effort statistics among programs is difficult. Catch rates may not reflect patterns of abundance, because they have been affected by regulatory changes (e.g., 1994 closed areas, 2000 trip limits, 2006 reductions in trip limits). However, evaluation of catch rates (kept + discarded) from observed tows that caught monkfish in the NFMA showed a peak in 2003 in the trawl fishery and in 2005 in the gillnet fishery, probably reflecting the strong 1999 yearclass. CPUE has since declined in the north (Figure A14). In the SFMA, CPUE indices have been relatively flat in the trawl and dredge fisheries for the past decade; however, gillnet indices increased steadily during 1999-2004, and have since held steady or declined slightly (Figure A14).

TOR 2. Report results of 2009 cooperative monkfish survey and describe sources of uncertainty in the data and results.

Methods - 2009 Monkfish Cooperative Survey

Survey Design and Protocols

The survey used a stratified random design with allocation proportional to stratum area (n=175 planned tows). An additional 35 tows (~17% of the total) were randomly selected in strata selected by industry members. In previous monkfish cooperative surveys (2001, 2004), sampling effort was allocated according to fishing effort patterns; however, this led to problems with interpretation of the 2004 survey which experienced extensive weather delays. Allocation of sampling effort using stratum area in 2009 addressed this concern and provided a basis for more direct comparison with the NEFSC 2009 spring survey conducted on the *FSV Henry Bigelow*.

Standard operating procedures were used on each vessel, including 30 minute tows (from time winches locked to time winches re-engaged for haul back) at 2.5 knots designated speed. Tow paths followed the depth contour. If pre-determined locations could not be sampled (due to fixed gear, bad bottom, etc.), stations were relocated as close as possible at a similar depth. A standard scope ratio of 2* tow depth plus 25 fathoms of wire was used for all nets.

The location of successful survey tows is shown in Figure A15. All survey tows were completed during Feb. 10 – Apr 26, 2010.

Ships and Gear

Two monkfish trawl vessels were contracted for the survey, both out of New Bedford. The *FV Endurance* (“ER”, 107 ft. stern trawler) sampled primarily the northern monkfish management area (U.S. waters of the Gulf of Maine and northern portion of Georges Bank) using two nets, one fitted with a cookie sweep for soft bottom, and one with roller gear for hard bottom (Figures A16 and A17). Both nets had a tickler chain (38 m of 3/8” chain). The *FV Mary*

K (“MK”, 96 ft. stern trawler) sampled in the southern management area (southern portion of Georges Bank and middle Atlantic Bight) using a net with a cookie sweep (Figure A18).

Sensor packages (Furuno on *Endurance*, NorthStar on *Mary K*) collected streams of data during each tow which included course over ground, speed over ground, GPS location (latitude, longitude), wingspread, bottom contact, depth and temperature. All types of data were not successfully collected for each tow. The number of tows with each type of sensor data is shown in Table A12 for each net type. Due to difficulties with obtaining wingspread measurements on the *Mary K* net, a set of dedicated mensuration tows were conducted to develop depth-wingspread relationships for the *Mary K*.

Analysis

Monkfish population estimates (biomass, numbers) were developed by estimating area swept during sampling in each stratum, converting this to monkfish density (kg, number caught per area swept), multiplying density by stratum area for each stratum, and summing over strata to derive total biomass and population size of monkfish in the two monkfish management areas. Population estimates were made using winch lock and winch re-engage to define tow duration (“nominal tow”) or using sensor data to define tow duration (“sensor tow”). Nominal and sensor tow population estimates were generated under different assumptions of net capture efficiency.

Area Swept Population Estimates

Area swept by each tow was calculated as

$$AS = TDis * WS$$

where

$$TDis = TDur * \overline{SOG}$$

and

AS = area swept (nmi^2)

TDis = distance covered by each tow in nmi

WS = wing spread in nmi

TDur = tow duration (nominal or sensor)

SOG = speed over ground during tow

To estimate population biomass and number, we calculated monkfish densities in each stratum as the sum of the numbers caught divided by the sum of the area swept. Biomass in each stratum was estimated as the product of number of fish and mean weight of fish in the stratum. Biomass and numbers were summed over strata to arrive at minimum biomass and population size. Biomass and population size were also estimated under two assumptions regarding net efficiencies.

$$N = \sum_h n_h$$

$$B = \sum_h n_h * \overline{w}_h$$

where

$$n_h = \left(\sum (n_i / c_j) / \sum a_i\right) * A_h$$

and

N= population size
B= biomass
n_h= number in stratum h
$\overline{w}_h$ = mean weight in stratum
i=tow number
c_j=efficiency of net j (proportion retained)
a_i=area swept during tow i
A_h=total area of stratum h

We used tows that had good quality sensor data to develop estimates of sensor tow data from nominal tow data, as follows:

To develop wingspread estimates for MK cookie, we applied a regression of wingspread against tow depth (Figure A19) developed from the mensuration experiments. Bottom contact readings were used to define the start of the tow, and winch re-engage (nominal stop time) was used to define the end of the tow; this generally coincided with tow end defined by bottom contact indicators because of the use of a separate winch engine on the Mary K. The deepest station for which we had wingspread measurements was 271 m. Approximately 13 % of stations were deeper than this (max. 480 m). Therefore we assumed a wingspread at 400 m equal to the average for tows greater than 200 m (n=4); this caused the predicted wingspread to decline at greater depths as would be expected (Weinberg and Kotwicki 2008).

A similar approach was used for ER tows that had no wingspread readings, except that bottom contact data were used to define the end of the tow as well as the beginning. For ER cookie, there were only 4 tows with both bottom contact and wingspread measurements, therefore we used wingspread during the nominal tow time to develop the depth-wingspread relationship (Figure A20). We used senor tow durations for the ER roller net, however, the relationship with depth was very similar to that derived from nominal tow times (Figure A20).

To develop tow duration for tows with no bottom contact sensor data, we adjusted tow duration according to relationships between depth and the relative difference between nominal and senor-defined tow durations (Figure A21). This relationship was relatively tight for the MK cookie sweep (r2=0.80), but much weaker and of smaller magnitude for the ER roller gear. Too few tows were available for the ER cookie sweep to estimate a relationship between nominal and sensor tow durations, so we applied the relationship for ER roller to ER cookie. The reason for the negative slope for MK cookie was that most sensor start times were after nominal start times, but sensor end times coincided with nominal end times, so sensor tows were generally shorter than nominal tows. For the ER, sensor start and end were both generally after nominal start and end (Appendix A2).

The following table summarizes the corrections applied to derive sensor tow durations and wingspread estimates for tows lacking sensor data.

Net	Wingspread predicted from	Sensor tow duration predicted from
MK Cookie	depth-wspread relation - MK cookie sensor data	depth-% difference relation - MK cookie sensor data
ER Cookie	depth-wspread relation - ER cookie nominal data	depth-% difference relation - ER roller sensor data
ER Roller	depth-wspread relation - ER roller sensor data	depth-% difference relation - ER roller sensor data

An additional adjustment was made to average tow speed for tows with no bottom contact data using relationships between nominal tow speed and tow speed during the sensor-defined tow period (Figure A22). This resulted in slower average tow speed during sensor-defined tows on the Endurance because speed dropped abruptly after winch lock, but bottom contact continued for a short period, thus bringing down the average speed for sensor tows. This pattern was not seen on the *Mary K*, which has an independent winch engine, thus nominal and sensor tow end occurred at the same time.

Net Efficiency

Depletion experiments were used to estimate efficiency of the 3 nets in capturing monkfish. The experiments were done by repeatedly towing over the same tow path, always in the same direction, until the monkfish catch approached zero. Eight depletion experiments were completed (4 for the *Mary K* cookie sweep, and 2 for each of the Endurance nets). The method used for data analysis is described in Rago et al. (2006). The location of the depletion experiments is shown in Figure A23.

Results

A total of 204 survey stations were successfully completed, and an additional 91 tows were made for depletion experiments and mensuration studies (Table A13). Figures A24-A26 show nominal catch rates (kg per tow, # per tow) for the survey stations. Figure A27 shows the depth distribution of sampling locations for survey tows.

Net Efficiency

The efficiency estimates derived from the depletion experiments are summarized in Table 14. For detailed description of the net efficiency analysis and results, see Appendix A1.
For three of the efficiency experiments, the estimation procedure was not successful (Appendix A.1) and the results were excluded from further analysis. Net efficiencies used to estimate population biomass and numbers were the average of experiments 1, 3, and 4 for the *Mary K* cookie sweep and experiments 5 and 7 for the *Endurance* cookie sweep. For the *Endurance* roller sweep, there were no successful experiments, so the results of experiments conducted during the 2001 cooperative survey comparing roller and cookie sweeps were used. These experiments found that the roller was 92% as efficient as the cookie sweep. We therefore used the average efficiency of the *Endurance* cookie sweep 0.249 * 0.92 = 0.229 as the efficiency of the 2009 net with roller gear. The efficiency estimates, called ‘intermediate’ in this report to correspond with earlier cooperative survey reports which additionally reported estimates based on a range (low and high) of efficiency estimates.

Population Estimates

Swept-area population point estimates are shown in Table A15 and Figure A29, and were on the order of 114-116 thousand mt (60-62 million fish) for the entire survey area assuming intermediate net efficiencies. Minimum estimates showed approximately 30% of the stock in the northern management area (which contains 42% of the survey area).

Differences between estimates derived from sensor tow durations were slightly higher (~8 %) than nominal estimates in the north and slightly lower (~6%) in the south (Table A15). In the north, the differences can be attributed to slower average speeds and shorter tow durations for sensor tows, which reduced the estimate of area swept and increased the estimate of density (Figure A28). In the south, adjustments to average speed and tow duration essentially cancelled each other, resulting in little difference in tow distance between nominal and senor estimates. Sensor-derived monkfish densities were lower than nominal densities because wingspread estimates were higher in sensor tows, thus increasing area swept and decreasing the density estimate (Figure A28).

The point estimates of area swept population size and biomass for 2009 are lower than those estimated from the 2001 survey (Table A15, Figure A29), with the exception of the south for efficiency-corrected and sensor-based estimates. (The 2004 survey is difficult to interpret due to extensive delays in completing the survey due to weather, but the 2001 survey is more comparable to the 2009 survey in that the two management areas were sampled simultaneously and the survey completed during Feb-April). The lower estimates for 2009 are driven by consistently lower densities (nominal # per nominal nmi swept) in the NFMA (Figure A30), which could be related to earlier start dates in that area than in 2001 (Table A15). In the south, there is no consistent difference between stratum densities in 2001 and 2009; however, the overall density is slightly lower in 2009 (Figure A31). Densities in the mid-Atlantic Bight (Hudson Canyon area and south) are higher in the deep water strata (greater than 200 fa) in 2009 than in the previous two surveys, suggesting that more monkfish may have been in deep water at the time of the 2009 cooperative survey.

In addition to density differences among years, the proportion of zero tows is higher in 2009 than in the earlier surveys (Table A15). This may be due in part to the change in allocation of sampling effort in 2009 (Figure A32).

The coefficient of variation developed by bootstrapping for the 2009 area swept population estimates was very low (Figure A33). This likely underestimates the true variance because of the relatively small number of tows in each stratum (and thus a small number to be drawn from in the bootstrapping).

Further bootstrapping analyses were used to compute the sampling distribution of biomass estimates in each management area from the 2001, 2004 and 2009 cooperative surveys using each of the valid depletion experiments within each year. Average monkfish density by management area was estimated from 1000 bootstrap samples. The distribution of efficiency estimates for each experiment was developed from 1000 bootstrap samples of the 95% confidence interval for the mean efficiency for each experiment. Each bootstrapped realization of density was divided by the corresponding bootstrapped efficiency estimate to develop 1000 estimates of population number, from which the mean and confidence intervals for each year, management area and experiment were derived. The estimated population numbers were converted to biomass using the mean fish weight for each year and management area. The resulting estimates are shown in Table A16.

Length, Age, Maturity

Expanded length frequencies from the cooperative survey (Figure A34) suggest a unimodal distribution in the north with the mode at around 35cm, and a bimodal distribution in the south with modes around 33 and 57 cm.

Samples were collected for aging studies but were not processed for this assessment due to uncertainty concerning validity of the aging method (NEFSC 2007a). However, a small number (n=25) of monkfish $\geq$ 80 cm were aged using the vertebral method for comparison with earlier samples (Figure A35).

Length-weight relationships for males and females from each management area are shown in Figure A36 and the parameters are listed in Table A17 along with parameters estimated from earlier studies. Maturation ogives are shown in Figure A37 and the parameters listed in Table A18 with estimates from earlier studies.

Comparison with NEFSC 2009 Spring Survey

The NEFSC spring survey was conducted during March 4 – May 8, 2010, generally proceeding from south to north. The spatial distribution of catches in the NEFSC survey was similar to catches from the cooperative surveys (Figure A38). Length frequencies from the NEFSC survey (Figure A39) reflect the gear's greater retention of smaller monkfish and lower overall catch rates (NEFSC total number of monkfish caught = 638, cooperative survey = 3,050). However, nominal minimum area swept estimates of biomass and population size were very similar for the northern area from the two surveys (Table A19). In the south, the estimates from the cooperative survey were approximately double those from the NEFSC survey for both biomass and population numbers.

Finding differences between results from the two surveys is not surprising because a number of operational characteristics differ. The NEFSC survey net has a codend liner with 1" mesh, while the cooperative survey nets used 6" mesh in the codend with no liner, thus the NEFSC survey captures smaller fish. The average tow speed was 3.1 kt during 20 minute tows (NEFSC) vs. 2.6 kt during 30-minute tows (Coop). Differences in net efficiency likely result from differences in the configuration of the net sweeps. In particular, the NEFSC survey net used roller gear for all tows whereas the cooperative survey net in the south used a cookie sweep which would be expected to tend bottom more closely and thus capture a higher proportion of the monkfish encountered. This may be important in the difference between surveys in estimates in the south. Finally, the cooperative survey sampled the southern Mid-Atlantic Bight in February, when monkfish are present across the shelf, while the Bigelow started a month later when monkfish have begun moving out of that area (Figure A40).

TOR 3. Characterize other survey data that are being used in the assessment (e.g., regional indices of abundance, recruitment, length data, state surveys). Describe the uncertainty in these sources of data.

Additional resource surveys used in the assessment include 2001 and 2004 cooperative monkfish surveys, NEFSC winter, spring and autumn offshore surveys, NEFSC scallop surveys (SFMA only), Northern Shrimp Technical Committee (NSTC) shrimp surveys (NFMA only), and ME/NH inshore surveys.

The NEFSC survey strata used to define the northern and southern management areas are:

Survey	Northern Area	Southern Area
NEFSC Offshore bottom trawl	20-30, 34-40	1-19, 61-76
NSTC Shrimp	1,3,5-8	
Shellfish		6,7,10,11,14,15,18,19,22-31,33-35,46,47,55,58-

NEFSC spring and autumn bottom trawl survey indices were standardized to adjust for statistically significant effects of trawl type (Sissenwine and Bowman 1977) on catch rates. The trawl conversion coefficients apply only to the spring survey during 1973-1981.

NEFSC indices derived from surveys on the FSV Henry Bigelow (starting spring 2009) were adjusted using calibration coefficients estimated during experimental work (Miller et al. 2009). The FSV *Henry B. Bigelow*, which became the main platform for NEFSC research surveys in spring 2009, has significantly different size, towing power, and fishing gear characteristics than the previous survey platform (*Albatross IV)*, resulting in different fishing power and catchability for most species. Calibration experiments to estimate these differences were conducted during 2008 (Brown 2009, NEFSC 2007b), and were peer reviewed by a Panel of three non-NMFS scientists during the summer of 2009 (Anonymous 2009). The objective was to develop specific protocols for guidance in the selection and use of appropriate estimators based on the amount of data available and the relative performance of two candidate estimators. The Panel developed general guidance on which estimator to use given sample sizes for each species. Following these guidelines, monkfish catches were converted using a simple ratio estimator without a seasonal (spring vs. fall) correction. The coefficients for monkfish were 7.1295 for numbers and 8.0618 for weight (kg) (Anonymous 2009; Miller et al. 2009).

Geographic distributions of survey catches are shown in Figures A40 to A42.

Northern Area

Indices from NEFSC autumn research trawl surveys indicate that biomass fluctuated without trend between 1963 and 1975, appears to have increased briefly in the late 1970's, but declined thereafter to near historic lows during the 1990's (Table A20, Figures A43 – A44). From 2000 to 2003, the index was greater than 2 kg/tow, but decreased to less than 1 kg/tow by 2008. Indices from the NEFSC spring research trawl surveys reflect similar trends of relatively high biomass levels in the mid 1970s (but with possible declines in the late 1970s), a declining trend from the early 1980s to the lowest values in the time series in 1998 an increase to relatively high biomass from 2001 to 2005, and somewhat lower levels since then (Table A21, Figures A43 and A45).

Abundance indices declined during the early 1960s, and then fluctuated without trend until the late 1980s. Abundance increased steadily from the late 1980s to a peak in 1994, declined during the late 1990s, and then peaked in 2000, reflecting a relatively strong 1999 yearclass. Abundance has declined steadily since 2000, but remains high relative to the earlier part of the time series.

Length distributions have become increasingly runcated over time (Figure A48). By 1990, fish greater than 60 cm long were uncommon in length frequency distributions. The minimum, median and maximum lengths in the trawl surveys declined steadily from the early 1980s until around 2000, when they began to increase again (Figure A49). Several modes potentially representing strong yearclasses have appeared consistently in survey distributions in recent years (Figures A48, A50).

Abundance indices were estimated for monkfish of lengths corresponding to ages 1 and 2 to help identify potential recruitment patterns (Figure A51). To the extent that these indices reflect recruitment, recruitment in the northern area has increased in the past decade. Relatively strong yearclasses were produced in 1993 and 1999. Survey abundance at age data (available since the mid 1990s) corroborates the suggestion of relatively strong 1993 and 1999 yearclasses in the northern area. Survey age data are available for 1993-2006 from the autumn trawl survey and for 1995-2006 for the spring trawl survey (NEFSC 2007a). Within the range of ages observed in the surveys, growth is essentially linear and there are no obvious differences with gender or management area. Other surveys which catch monkfish in the northern area include the ASMFC shrimp survey, the Massachusetts Division of Marine Fisheries fall and spring surveys, and ME/NH inshore surveys. These surveys sample only a portion of the stock area and may be affected by inconsistent coverage over time.

The shrimp survey samples the western Gulf of Maine during summer and caught more monkfish than the spring or fall surveys prior to 2009 (when the FSV Bigelow survey series began) (Table A22, Figures A43 and A46). Patterns of abundance and biomass have been relatively consistent among the spring, fall and shrimp surveys (NEFSC 2007a). The Massachusetts surveys catch few monkfish and were not considered to reflect patterns of abundance for the entire management area; therefore are not reported in the assessment (NEFSC 2007a). ME/NH inshore surveys began in 2000 and are conducted in spring and fall (Figure A47). Indices show similar trends to those from NEFSC and shrimp surveys (Table A23, Figure A43 and A.46).

Southern Area

Biomass indices from the NEFSC autumn research survey were high during the mid-1960s, fluctuated around an intermediate level during the 1970s-mid 1980s, then declined to consistently low levels since the late 1980s (Table A24, Figures A52 and A53). The biomass index increased slightly above the existing biomass threshold in 2001 and has been relatively stable, or declining slightly since then. NEFSC spring surveys reflect similar trends as the autumn series: biomass remained fairly high during the mid 1970s - early 1980s, but fluctuated around lower levels thereafter (Table A25, Figures A52 and A54). A spike in biomass was observed in 2003, but subsequent indices have returned to lower values. Biomass and abundance indices based on the NEFSC winter flatfish survey (conducted during 1992-2007) fluctuated without trend (Table A26, Figures A52 and A55). Although the winter survey series had a short duration, the gear used in the winter survey was more effective for capturing monkfish than the gear used in autumn or spring surveys. Abundance indices based on the NEFSC sea scallop survey show an increasing trend during 1984-1994 followed by a rapid decline from 1994-1998 and fluctuations around a relatively level during 2006-200 (Table A27, Figure A56).

Inconsistent geographic coverage should be considered in the interpretation of southern survey indices. For example the fall survey did not sample southern strata until 1967. The winter survey sampled Georges Bank inconsistently and did not sample deep strata before 1998. The scallop survey does not currently sample the entire southern management area.

Abundance (numbers per tow) shows trends similar to biomass, with a spike in 1972, fluctuations around a relatively low level since the mid-1970s, a slight increase in 2002 and 2003 followed by a return to lower levels. Length distributions from the southern area showed increasing truncation over time, but the size distribution appears to have stabilized in recent years (Figure A57). Maximum lengths declined by approximately 20 cm or more over the time series (Figure A58). As in the northern area, fish greater than 60 cm have been rare since the 1980s, especially when compared to the 1960s. Any recent strong recruitment does not appear to survive long enough to contribute substantially to increased stock biomass. Survey age data are available for 1993-2006 from the autumn trawl survey, 1995-2006 for the spring trawl survey and 1997-2007 for the winter trawl survey (NEFSC 2007a). Age samples collected since the 2006 survey have not been processed due to uncertainties regarding validity of the aging method (NEFSC 2007a).

Combined Management Areas

Survey indices for combined management areas for spring and fall are shown in Table A28 and A29, and Figures A59 – A61. Length composition trends are shown in Figures A62-A63.

TOR 4. Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series, and characterize the uncertainty of those estimates.

Several candidate modeling approaches were investigated by the Data Poort Stocks Working Group (NEFSC 2007a), but the only one considered suitable was a relatively new approach called SCALE (for Statistical Catch-At-Length Analysis). Results from this model were used in 2007 to estimate fishing mortality, recruitment and stock biomass and to redefine reference points. The SCALE model was updated and serves as the primary basis for the current assessment.

Monkfish SCALE Model

Introduction

Incomplete or lacking age-specific catch data and survey indices often limit the application of a full age-structured assessment (e.g. Virtual Population Analysis and many forward projecting age-structured models). Stock assessments often rely on the simpler size/age aggregated models (e.g. surplus production models) when age-specific information is lacking. However these models may not utilize all of the available information for a stock assessment. Knowledge of a species growth and lifespan, along with total catch data, size composition of the removals, recruitment indices and indices on numbers and size composition of the recruited fish in a survey can provide insights on population status using a simple model framework.

The Statistical Catch At Length (SCALE) model, is a forward projecting age-structured model tuned with total catch (mt), catch at length or proportional catch at length, recruitment at a specified age (usually estimated from first length mode in the survey), survey indices of abundance of the larger/older fish (usually adult fish) and the survey length frequency distributions. The SCALE model was developed in the AD model builder framework. The model parameter estimates are fishing mortality and recruitment in each year, fishing mortality to produce the initial population (Fstart), logistic selectivity parameters for each year or blocks of years and Qs for each survey index.

The SCALE model was developed as an age-structured model that does not rely on age-specific information on a yearly basis. The model is designed to fit length information, abundance indices, and recruitment at age which can be estimated by using survey length slicing. However the model does require an accurate representation of the average overall growth of the population which is input to the model as mean lengths at age. Growth can be modeled as sex-specific growth and natural mortality or growth and natural mortality can be modeled with the sexes combined. The SCALE model will allow for missing data.

Model Configuration

The SCALE model assumes growth follows the mean input length at age with predetermined input error in length at age. Therefore a growth model or estimates of the average mean length at age is essential for reliable results. The model assumes static growth and therefore population mean length/weight at age are assumed constant over time.

The SCALE model estimates logistic parameters for a flattop selectivity curve at length in each time block specified by the user for the calculation of population and catch age-length matrices or the user can input fixed logistic selectivity parameters. Presently the SCALE model cannot accommodate a dome shaped selectivity pattern.

The SCALE model computes an initial age-length population matrix in year one of the model as follows. First the estimated populations numbers at age starting with age-1 recruitment are normally distributed at 1 cm length intervals using mean length at age with the assumed standard deviation. Next the initial population numbers at age are calculated from the previous age at length abundance using the survival equation. An estimated fishing mortality (Fstart) is also used to produce the initial population. This F can be thought of as the average fishing mortality that occurred before the first year in the model. Now the process repeats itself with the total estimated abundance at age being redistributed according to the mean length at age and standard deviation in the next age (age+1).

This two step process is used to incorporate the effects of length specific selectivities and fishing mortality. The initial population length and age distribution is constructed by assuming population equilibrium with an initial value of F, called F_{start}. Length specific mortality is estimated as a two step process in which the population is first decremented for the length specific effects of mortality as follows:

$$N^{*}_{a,len,y_1} = N_{a-1,len,y_1} e^{-\left(PR_{len} F_{start} + M\right)}$$

In the second step, the total population of survivors is then redistributed over the lengths at age *a* by assuming that the proportions of numbers at length at age *a* follow a normal distribution with a mean length derived from the input growth curve (mean lengths at age).

$$N_{a,len,y_1} = \pi_{len,a} \sum_{len=0}^{L_\infty} N^{*}_{a,len,y_1}$$

where

$$\pi_{len,a} = \Phi\left(len+1 \mid \mu_a, \sigma_a^2\right) - \Phi\left(len \mid \mu_a, \sigma_a^2\right)$$

where

$$\mu_a = L_\infty\left(1 - e^{-K(a-t_0)}\right)$$

Mean lengths at age can be calculated from a von Bertalanffy model from a prior study as shown in the equation above or mean lengths at age can be calculated directly from an age-length key. Variation in length at age a = • $_s^2$ can often be approximated empirically from the growth study used for the estimation of mean lengths at age. If large differences in growth exist between the sexes then growth can be input as sex-specific growth with sex-specific natural mortality. However catch and survey data are still fitted with sexes combined.

This SCALE model formulation does not explicitly track the dynamics of length groups across age because the consequences of differential survival at length at age a do not alter the mean length of fish at age a+1. However, it does realistically account for the variations in age-specific partial recruitment patterns by incorporating the expected distribution of lengths at age.

In the next step the population numbers at age and length for years after the calculation of the initial population use the previous age and year for the estimate of abundance. Here the calculations are done on a cohort basis. As in the previous initial population survival equation, the partial recruitment is estimated on a length vector.

$$N^*_{a,len,y} = N_{a-1,len,y-1}e^{-\left(PR_{len}F_{y-1}+M\right)}$$

second stage

$$N_{a,len,y} = \pi_{len,a}\sum_{len=0}^{L_\infty} N^*_{a,len,y}$$

Constant M is assumed along with an estimated length-weight relationship to convert estimated catch in numbers to catch in weight. The standard Baranov=s catch equation is used to remove the catch from the population in estimating fishing mortality.

$$C_{y,a,len} = \frac{N_{y,a,len}\,F_y\,PR_{len}\left(1 - e^{-\left(F_y PR_{len}+M\right)}\right)}{(F_y \quad PR_{len}) + M}$$

Catch is converted to yield by assuming a time invariant average weight at length.

$$Y_{y,a,len} = C_{y,a,len}W_{len}$$

The SCALE model results in the calculation of population and catch age-length matrices for the starting population and then for each year thereafter. The model is programmed to estimate recruitment in year 1 and estimate variation in recruitment relative to recruitment in year 1 for each year thereafter. Estimated recruitment in year one can be thought of as the estimated average long term recruitment in the population since it produces the initial population. The residual sum of squares of the variation in recruitment • $(\text{Vrec})^2$ is then used as a component of the total objective function. The weight on the recruitment variation component of the objective function (Vrec) can be used to penalize the model for estimating large changes in recruitment relative to estimated recruitment in year one.

The model requires an age-1 recruitment index for tuning or the user can assume relatively constant recruitment over time by using a high weight on Vrec. Usually there is little overlap in ages at length for fish that are one and/or two years of age in a survey of abundance. The first mode in a survey can generally index age-1 recruitment using length slicing. In addition numbers and the length frequency of the larger fish (adult fish) in a survey where overlap in ages at a particular length occurs can be used for tuning population abundance. The model tunes to the catch and survey length frequency data using a multinomial distribution. The user specifies the minimum size (cm) for the model to fit. Different minimum sizes can be fit for the catch and survey data length frequencies.

The number of parameters estimated is equal to the number of years in estimating F and recruitment plus one for the F to produce the initial population (Fstart), logistic selectivity parameters for each year or blocks of years, and for each survey Q. The total likelihood function to be minimized is made up of likelihood components comprised of fits to the catch, catch length frequencies, the recruitment variation penalty, each recruitment index, each adult index, and adult survey length frequencies:

$$\mathrm{L}_{\mathrm{catch}} = \sum_{years}\left(l\,\mathrm{n}(Y_{\mathrm{obs,y}} + 1) - \ln\left(\sum_{a}\sum_{len} \mathrm{Y}_{\mathrm{pred,len,a,y}} + 1 \right)\right)^2$$

$$L_{catch\_lf} = -N_{eff} \sum_{y} \left(\sum_{inlen}^{L_\infty} \left(\left(C_{y,len} + 1\right) \ln\left(1 + \sum_{a} C_{pred,y,a,len} \right) - \ln\left(C_{y,len} + 1\right) \right) \right)$$

$$\mathrm{L}_{\mathrm{vrec}} = \sum_{y=2}^{Nyears} \left(Vrec_y\right)^2 = \sum_{y=2}^{Nyears} \left(R_1 - R_y\right)^2$$

$$\sum L_{rec} = \sum_{i=1}^{Nrec} \left[\sum_{y}^{Nyears} \left(\ln\left(I_{rec_i,inage_i,y}\right) - \ln\left(\sum_{len}^{L_\infty} N_{y,inage_i,len} * q_{rec\,i} \right) \right)^2 \right]$$

$$\sum L_{adult} = \sum_{i=1}^{Nadult} \left[\sum_{y}^{Nyears} \left(\ln\left(I_{adult_i,inlen+_i,y}\right) - \left(\sum_{a} \sum_{inlen_i}^{L_\infty} \ln\left(N_{pred,y,a,len} * q_{adult\,i}\right) \right) \right)^2 \right]$$

$$\sum L_{lf} = \sum_{i=1}^{Nlf} \left[-N_{eff} \sum_{y} \left(\sum_{inlen_i}^{L_\infty} \left(\left(I_{lf_i,y,len} + 1\right) \ln\left(1 + \sum_{a} N_{pred,y,a,len} \right) - \ln\left(I_{lf_i,y,len} + 1\right) \right) \right) \right]$$

In equation L_{catch_lf} calculation of the sum of length is made from the user input specified catch length to the maximum length for fitting the catch. Input user specified fits are indicated with the prefix "in" in the equations. LF indicates fits to length frequencies. In equation L_{rec} the input specified recruitment age and in L_{adult} and L_{lf} the input survey specified lengths up to the maximum length is used in the calculation.

$$Obj\ \ fcn = \sum_{i=1}^{N} \lambda_i L_i$$

Lambdas represent the weights to be set by the user for each likelihood component in the total objective function.

Monkfish SCALE Model Configuration and Results

No new information on growth and natural mortality exists for this assessment. Growth, variation in mean length at age, and natural mortality (M=0.3) did not change from the assumptions used in the 2007 assessment (NEFSC 2007a). Mean and variance in monkfish length at age were estimated from industry-based surveys (2001 and 2004), and NEFSC winter, spring, and fall surveys for management areas combined (Table A30). No significant differences in growth were observed between the management units in the 2001 and 2004 cooperative surveys. The standard deviation for age 1 was 2.9; for older ages a standard deviation of 4.5 was assumed. The overall standard deviation on mean lengths at age was estimated directly from the age data. The oldest aged fish from surveys and commercial samples was age 12. Mean lengths at age for the older fish (10-12) was supplemented with data collected from a study of large monkfish (Johnson et al. 2008).

Age modes in the predicted length frequencies are seen for most ages due to the linear nature of monkfish growth and the model structure that uses a single annual growth time step (Appendix A1). The absence of a decline in growth with age in monkfish produces this process error in the SCALE model fits. This can be concealed by increasing the variance on mean lengths at age by increasing the assumed variance on the mean lengths at age. However, as in the 2007 assessment, an increase in the variance on the mean lengths at age beyond what is supported by the raw growth data was not done due to concerns on its effect on the estimated selectivity.

Relative abundance trends for recruits (ages 1, 2, and/or 3) and adults (40+ cm) in each management unit were updated and are shown in Figures A64 through A69. The length interval specific to each survey used as a proxy for the recruitment ages are shown in the plots. For both management units, the model was fit to spring, fall and industry-based survey length frequencies (30+ cm), 40+ cm adult indices, and recruitment indices at age. The northern area had additional inputs from a shrimp trawl survey (1991-2009) and the southern area used the NEFSC winter trawl (1992-2007) and NEFSC scallop dredge (1984-2009) surveys. Inputs from the fall inshore ME/NH trawl survey (2000-2009) were added to the northern management area in this assessment (Figures A70 and A71). The use of the Fall MDMF bottom trawl survey was also investigated in this assessment but was dropped as an index of abundance (Figure A72). The working group concluded that this index was unreliable for monkfish due to the low numbers of fish caught in the survey.

Indices at age and adult 40+ cm abundance indices were scaled using the approximate area (nm^2) of the survey divided by the average coverage of the survey's tow (Table A31). The survey catchability estimates from the model were used as a diagnostic check for the interpretation of survey efficiencies. Survey indices from the R/V Bigelow were converted to Albatross units for 2009 (numbers per tow / 7.2). An additional diagnostic run for each management area (north, south and combined) that included the absolute estimates of the cooperative monkfish 40+ cm estimates for all three years was investigated. An assumed 50 percent efficiently was used for the 2009 cooperative monkfish survey. The estimated q's from the model for the cooperative monkfish survey ranged from 0.68 to 1.18 but the model could not fit the large fluctuation in abundance between survey years (Figure A73).

There is no evidence of strong recruitment in the age-specific indices over the last three years (2007-2009). The 40+ cm indices also indicate a decline in abundance in comparison to the previous three years. There was little change in the survey and catch length frequency distributions since the 2007 assessment (Appendix A1).

In the 2007 assessment a single selectivity block (1980-2009) was estimated for the northern management unit and three selectivity blocks were estimated for the southern management unit. A single selectivity block for the north was retained for this assessment. Shifting the second selectivity block from 2003-2004 (2007 assessment) to 2001-2002 (current assessment) in the south provided a better fit to the catch length frequency data and corresponded better to the shift to gillnet gear in the fishery. The first selectivity block in the southern area (1980-1995) that was established in the 2007 assessment has only two years of length information and appears to produce unstable selectivity estimates in this assessment, therefore it was eliminated in the final southern run 8.

For the 2007 assessment a variety of conditions and assumptions were tested using sensitivity runs and a similar approach was taken for SARC 50. Comparisons of the configuration and results of the final and sensitivity SCALE runs for this assessment are shown in Tables A32 through A34 and Figures A74 through A80. The influence of three additional years of data to the final configuration of the 2007 assessment was determined in run 1 in both the north and southern management areas. In the north run 2 determined the influence of adding both the ME/NH survey and the MA DMF survey. In runs 3 and greater the MDMF survey was dropped from the model. The model was allowed to estimate Fstart in runs 4 to 7 and runs 6 and 7 were done to test sensitivity to the Vrec (recruitment variation) penalty weight. In the south, runs 2 to 7 allowed estimation of Fstart; runs 3 to 5 also tested alternative selectivity blocks.

Similar to the 2007 assessment, models for both the north and south had difficulty in fitting the catch length frequency data in the last few years. Fits to the catch length frequencies can be seen Appendix A1. A significant decline in the catch has occurred in the last three years of the model. However there is no evidence of an increase in the number of larger fish in the catch or in any of the survey length frequency distributions from 2007 to 2009. The model could not reconcile the effects of a decline in catch with the lack of a corresponding shift in the length distributions. Sensitivity run 5 in the north and runs 6 and 7 puts higher weight on the length distributions in the model. This resulted in a lack of fit to the catch (Figure A80).

The sensitivity runs of the SCALE model produced similar trends in F and biomass. As in the 2007 assessment the trade-off between shifts in the estimated selectivity and other weighting components of the model still exist.

Combining the northern and southern areas into a single assessment model was investigated in this assessment. In general the combined assessment model results were intermediate between the northern and southern model runs (Figure A79). Combined biomass estimates approximated the sum for the two area runs.

The final working group model runs retained for the 2007 assessment assumed fixed parameters for Fstart (North at 0.01, South at 0.2). The northern area results suggested there were at least two strong recruitment pulses during the 1990s that fueled subsequent increases in the catch (Figures A75 and A80). These strong recruitment events were not evident in the south (Figures A78 and A80). The final northern run estimated lower abundance with a shift in selectivity to larger fish relative to the 2007 assessment. The northern final model estimated much lower abundance in the terminal year than what was projected from the 2007 assessment; 144,000 tons in 2007 versus 66,000 tons in the current assessment (Figure A75). The final model for the southern area estimated relatively low recruitment in the last five years (2005-2009) of the model. However biomass and F predictions were similar to estimates from the 2007 assessment. Recruitment, biomass and fishing mortality estimates from the current assessment final runs are listed in Table A35.

The estimates of total biomass from the SCALE model fall within the confidence intervals (25^{th}-75^{th} percentile) of biomass estimates from the cooperative surveys for 2001 and 2004 (Table A16); however, the 2009 estimates from the SCALE model are approximately double the absolute biomass estimates from the cooperative survey for 2009. The effect of the retrospective pattern in the SCALE estimates has not been factored into these comparisons.

Monkfish SCALE model Uncertainty

Assessment of monkfish is difficult because of the often-poor quality of data available. Survey data provide a long-term picture, but there is high variability in the survey trends due to the low numbers of fish caught in many of the surveys. Landings were historically under-reported and discard data were not available until relatively recently. Age samples were not taken in surveys until 1994 and from landings until 2000, and the landings are sparsely sampled for age even at present because removing vertebrae compromises product quality. Important aspects of monkfish biology are poorly understood, including stock structure and movement patterns, growth rates and longevity. Ageing methods have not been validated using known-age individuals. Effects of the process error within the model due to the linear growth trend are unknown. There is uncertainty surrounding the lack of an explanation for the consistent sex ratio patterns that occur with size in multiple surveys (Richards et al., 2008).

Given the litany of data limitations, it is not surprising that most of the assessment approaches applied were not successful during the 2007 Data Poor Stocks Working Group assessment. The SCALE model was considered useful at that assessment because it integrated the available information and the resulting estimates appeared reasonable (e.g. biomass estimates consistent with empirically-estimated biomass from industry-based surveys). This is still true in the current assessment. However, in this assessment substantial uncertainty remains surrounding the lack of evidence for rebuilding of the size structure with the observed decline in the catch.

Retrospective analyses suggest there is higher uncertainty with the northern management model relative to the southern management assessment (Figures A81 and A82). The northern model exhibits strong retrospective patterns in fishing mortality and stock size. If the fishing mortality estimated for 2009 is adjusted upward to account for the average retrospective underestimation of -66% for the 2002-2008 terminal years, the estimate for 2009 changes from 0.10 to 0.17. If the total biomass estimated for 2009 is adjusted downward to account for the average retrospective overestimation of +108% for the 2002-2008 terminal years, the estimate for 2009 changes from 66,062 mt to 31,761 mt. The model for the southern area exhibits moderate retrospective patterns in fishing mortality and stock size. If the fishing mortality estimated for 2009 is adjusted upward to account for the average retrospective underestimation of -13% for the 2002-2008 terminal years, the estimate for 2009 changes from 0.07 to 0.08. If the total biomass estimated for 2009 is adjusted downward to account for the average retrospective overestimation of +16% for the 2002-2008 terminal years, the estimate for 2009 changes from 131,218 mt to 113,119 mt. The model for the combined area exhibits intermediate retrospective patterns in fishing mortality and stock size with respect to the separate areas (Figure A83). Age specific retrospective adjustments using seven peels are summarized in Table A36.

Potential explanations for the lack of fit and/or retrospective pattern in the SCALE model are summarized in Table A37. The explanations deemed most likely to cause underlying problems with the model were (1) the growth model is incorrect (ie. growth is not linear with age) and (2) setting M=0.3 is inappropriate (ie. monkfish longevity may be greater than currently assumed).

Improvements to the SCALE model allow for estimation of within model uncertainty on fishery selectivity and stock numbers through the MCMC procedure. However, uncertainty in F could not be estimated with the MCMC for monkfish because fishing mortality is set equal to model results in the MCMC. Therefore all of the within model uncertainty is not accounted for in the MCMC results. The high uncertainty surrounding this assessment will be largely underestimated by within model uncertainty estimates and probably should not be solely used for the determination of the uncertainty in setting ABCs. As in the 2007 assessment, the results are dependent on the input mean lengths at age as an appropriate approximation for monkfish growth.

Spawning biomass is not output directly by the SCALE model, but was estimated as the product of population numbers at length (SCALE), maturity at length (Richards et al. 2008), weight at length (SCALE) and fraction female at length (based on data in Richards et al. 2008). The fraction female at length was estimated two ways: (1) using observed patterns of proportion female vs. length in the south and north (e.g. Richards et al. 2008) and (2) assuming sex ratio=50:50 up to 70 cm, then 100% female for fish $\geq$ 70 cm. Ogives were averaged to develop estimates for the combined stock areas. Trends in spawning biomass are shown in Figure A84.

SAW50 Editor's note: The SARC50 panel acknowledged the high degree of uncertainty in estimates from the SCALE model due to data limitations, poorly understood monkfish biology (growth, natural mortality, stock structure), and the strong retrospective pattern in the northern area. The panel did not favor directly adjusting for the retrospective pattern. Despite the high uncertainty, the model was accepted, but with strong precautionary caveats.

TOR 5. Update or redefine biological reference points (BRPs; estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, and F_{MSY}; and estimates of their uncertainty). Comment on the scientific adequacy of existing and redefined BRPs.

Overfishing Reference Points

SAW 34 (NEFSC 2002) and Framework 2 of the Monkfish FMP established the overfishing definition as F_{max} and estimated it be equal to 0.2 for both management areas (assuming M=0.2). NEFSC (2007a) examined length-based and age-based YPR models and concluded that the length-based approach was not appropriate as it assumes a von Bertalanffy growth model which does not fit currently understood monkfish growth patterns. NEFSC (2007a) used the age-based YPR model to update the value of Fmax assuming M=0.3 and the current assessment updates this model again using revised selectivity patterns output from SCALE. F_{target} was not defined in the original monkfish FMP or in Framework Adjustment 2. The DPSWG (NEFSC 2007a) recommended that $F_{40\%}$ be used to define F_{target}.

Age-based YPR was calculated for each management region using the approach of NEFSC (2007a). This assumed a constant natural mortality M=0.3 and applied selectivity at age approximated from SCALE output selectivity at length for each area. Mean weights at age for the catch and stock were from SCALE output, and maturity ogives were from 2001 Cooperative Monkfish Survey data (NEFSC 2002), which were very similar to other estimates of maturity (Table A18, Figure A85). The estimates from NEFSC (2007a) and the current assessment are shown in Table A38. The difference in estimates for the two areas reflects differing selectivity of gillnets and trawls; more monkfish are landed using gillnets in the south than in the north. The differences between years reflect the changes in selectivity patterns estimated by the SCALE model.

Biomass reference points

Biomass reference points were developed by NEFSC (2007a) using results of the SCALE model. The recommended $B_{threshold}$ was the lowest observed value in the total biomass time series (1980-present) from which the stock has then increased (termed "B_{Loss}"), estimated in 2006 to be 65,000 mt in the north and 96,000 mt in the south. The recommended B_{target} was the average of total biomass for the time period (1980-present), estimated in 2006 to be 92,000 mt in the north and 123,000 mt in the south.

The 2010 assessment updated biomass reference points developed by NEFSC (2007a) based on results of the 2009 SCALE population model (Table A39). Using the current FMP definitions, updated estimates of $B_{threshold}$ are 41,238 mt of total stock biomass in the northern area and 99,181 mt in the southern area. Estimates of B_{target} (average of 1980-2006 estimates) are 62,371 mt of total stock biomass in the northern area and 120,292 mt in the southern area. Biomass reference points for the combined areas approximated the sum for the two existing management areas (i.e., relative scaling persisted). Using the current FMP definitions, the combined area estimate of $B_{threshold}$ is 159,715 mt (average of 1980-2009 estimates) and the combined area estimate of B_{target} (average of 1980-2009 estimates) is 208,190 mt.

Biomass reference points for New England groundfish stocks have recently been based on the long-term projected biomass corresponding to F_{MSY} or its proxy, which for monkfish would be F_{max}. In keeping with this practice, proposed total biomass targets (i.e., B_{max} at F_{max}) and thresholds ($0.5*B_{max}$) were calculated for monkfish for the northern, southern and combined areas (Table A39). Using this approach, proposed estimates of B_{target} are 52,930 mt in the northern area and 74,490 mt in the southern area, and estimates of $B_{threshold}$ are 26,465 mt in the northern area and 37,245 mt in the southern area. The combined area estimate of B_{target} 129,002 mt and the estimate of $B_{threshold}$ is 64,501 mt. The total catch produced from the long-term Btarget at the respective values of F_{max} (i.e., proxy for F_{MSY}), is 10,745 mt for the northern area, 15,279 mt for the southern area, and 25,943 mt for the areas combined.

All of the BRPs are based on results of the SCALE model (including F reference points from the YPR which uses selectivity curves estimated by SCALE), therefore the BRPs are subject to the same high level of uncertainty that surrounds the SCALE model results. The BRPs developed by NEFSC (2007a) were *ad hoc* and are problematic in that BRPs change with every update or modification of the model. Further, the results for the southern management area indicate that biomass approached overfished status in the mid-1990s even though F remained below F_{target}. This suggests that those BRPs were unreliable. The BRPs based on projected biomass at F_{max} are also subject to high uncertainty due to reliance on projections of SCALE model results and the high estimate of F_{max} due to the assumption of M=0.3 in the YPR model. The biomass reference points using the current method are much lower, which accounts for the more optimistic view of stock size relative to the biomass target and biomass threshold.

SAW50 Editor's note: The SARC50 panel recommended adoption of the biomass reference points based on "Bmax projected".

TOR 6. Evaluate stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 5).

Based on the existing biological reference points from the 2007 stock assessment and the Monkfish Fishery Management Plan (FMP), monkfish would be considered not overfished with no overfishing occurring for both the northern and southern stock management areas (Figure A86, Table A39). In the northern area, the existing $B_{threshold}$ is 65,200 mt of total stock biomass and the existing $F_{threshold}$ is F_{max} = 0.31. The estimated 2009 northern area biomass is 66,062 mt, above the existing $B_{threshold}$; the estimated northern area F in 2009 is 0.10, below the existing $F_{threshold}$. In the southern area, the existing $B_{threshold}$ is 96,400 mt and the existing $F_{threshold}$ is F_{max} = 0.40. The estimated 2009 southern area biomass is 131,218 mt, above the existing $B_{threshold}$; the estimated southern area F in 2009 is 0.07, below the existing $F_{threshold}$.

The 2010 assessment has updated the biological reference points based on an updated yield-per-recruit analysis and the results of the SCALE length-tuned population model that incorporates multiple survey indices and catch data. Based on proposed reference points from these updated analyses, monkfish in both management areas are not overfished with no overfishing occurring (Figure A87). Using the current FMP definitions, updated estimates of

$B_{threshold}$ are 41,238 mt of total stock biomass in the northern area and 99,181 mt in the southern area. Estimates of B_{target} (average of 1980-2006 estimates) are 62,371 mt in the northern area and 120,292 mt in the southern area. Estimates of total biomass for 2009 are 66,062 mt in the northern area and 131,218 mt in the southern area, above B_{target} for both areas. The existing overfishing threshold is based on F_{max}, and this was retained in the 2010 assessment. The updated estimates of F_{max} are 0.43 per year in the northern area and 0.46 per year in the southern area. Estimates of current F (2009) are 0.10 per year in the northern area and 0.07 per year in the southern area, both less than the respective overfishing thresholds.

A combined stock area model was constructed to address a Research Recommendation from the 2007 assessment. Biomass reference points for the combined areas approximated the sum for the two existing management areas (i.e., relative scaling persisted). Using the current FMP definitions, the combined area estimate of $B_{threshold}$ is 159,715 mt of total stock biomass (average of 1980-2009 estimates) and the combined area estimate of B_{target} (average of 1980-2009 estimates) is 208,190 mt. The estimate of combined area total biomass for 2009 is 255,326 mt, above B_{target}. The combined area overfishing threshold based on F_{max} is 0.37. The combined area estimate of current F (2009) is 0.05, below the combined area overfishing threshold (Figure A88).

Biomass reference points for New England groundfish stocks have recently been based on the long-term projected biomass corresponding to F_{MSY} or its proxy, which for monkfish would be F_{max}. In keeping with this practice, proposed total biomass targets (i.e., B_{max} at F_{max}) and thresholds ($0.5*B_{max}$) were calculated for monkfish for the northern, southern and combined areas. Using this approach, proposed estimates of B_{target} are 52,930 mt in the northern area and 74,490 mt in the southern area, and estimates of $B_{threshold}$ are 26,465 mt in the northern area and 37,245 mt in the southern area (Table A39, Figure A89). The combined area estimate of B_{target} 129,002 mt and the estimate of $B_{threshold}$ is 64,501 mt. The total catch produced from the long-term B_{target} at the respective values of F_{max} (i.e., proxy for F_{MSY}), is 10,745 mt for the northern area, 15,279 mt for the southern area, and 25,943 mt for the areas combined.

The assessment results for monkfish continue to be uncertain due to likely under-reported landings and unknown discards during the 1980s and incomplete understanding of key biological parameters such as age and growth, longevity, natural mortality and stock structure. The population models for all areas exhibit retrospective patterns that are strongest for the 2002-2006 terminal years and weaker for the 2007-2008 terminal years. The retrospective patterns are strongest for the northern area, weakest for the southern area, and intermediate for the model of combined areas (Figures A81-A83). The BRPs are all based on output from the SCALE model, therefore the BRPs are also highly uncertain.

TOR 7. Evaluate monkfish diet composition data and its implications for population level consumption by monkfish.

Food habits were evaluated for monkfish as major a predator in the ecosystem. The total amount of food eaten and the type of food eaten were the primary food habits data examined. From these basic food habits data, diet composition, per capita consumption, total consumption, and the amount of prey removed by monkfish were calculated. Contrasts to total energy flows in the ecosystem and fishery removals of commercially targeted skate prey were conducted to fully address the Term of Reference.

Methods

To estimate mean stomach contents (S_i), the total amount of food eaten (as observed from food habits sampling) was calculated for each size class, temporal and/or spatial scheme. The denominator in the mean stomach contents (i.e., the number of stomachs sampled) was inclusive of empty stomachs. These means were weighted by the number of tows in a temporal and spatial scheme as part of a two-stage cluster design. Further background on food habits sampling protocols and these estimators can be found in Link and Almeida (2000). This sampling program was a part of the NEFSC bottom trawl survey program (Azarovitz 1981; NEFC 1988). Units are in g.

Estimates were calculated on an annual basis for each monkfish size class, temporal and spatial combination. The size classes were < and• 40 cm for Small (S) and Large (L) size classes, respectively and the areas were southern and northern management regions. Although the food habits data collections started quantitatively in 1973, collections for monkfish weren't initiated until 1977. Key diagnostics were the number of empty stomachs over time and mean length vs. mean stomach contents weight (with ± 95% CI), which were examined to identify any major outliers in the data and to ascertain any notable patterns in variance.

To estimate diet composition (D_{ij}), the amount of each prey item was summed across all monkfish stomachs. These estimates were then divided by the total amount of food eaten in a size class, temporal and spatial scheme, totaling 100%. These estimates are proportions and were only presented for those major prey comprising >85% of the total for each size class, temporal and spatial scheme.

The approach to calculating consumption followed previously established methods, using an evacuation rate model methodology. For further details, see Durbin et al. (1983), Ursin et al. (1985), Pennington (1985), Overholtz et al. (1991, 1999, 2000, 2008), Tsou & Collie (2001a, 2001b), Link & Garrison (2002), Link et al. (2002, 2006, 2008, 2009), Link & Sosebee (2008), Overholtz & Link (2007), Tyrrell et al. (2007, 2008), Link and Idoine (2009), Moustahfid et al. (2009a, 2009b), and NEFSC (2006, 2007a, 2007b, 2008). The main data inputs are mean stomach contents (S_i) for each monkfish size-time-space scheme i, diet composition (D_{ij}) where j is the specific prey of interest, and T is the bottom temperature taken from the bottom trawl surveys (Taylor et al. 2005). Estimates of variance about all input variables were calculated.

Using the evacuation rate model to calculate consumption requires two variables and two parameters. The per capita consumption rate, C_i is calculated as:

$$C_i = 24 \cdot E_i \cdot \overline{S_i}^{\gamma} \quad ,$$

where 24 is the number of hours in a day and the evacuation rate E_i is:

$$E_i = \alpha e^{\beta T} \quad ;$$

and is formulated such that estimates of mean stomach contents (S_i) and ambient temperature (T; here used as bottom temperature from the NEFSC bottom trawl surveys (Taylor et al. 2005)) are the only data required. The parameters • and • are set as values chosen from the literature (Tsou and Collie 2001a, 2001b, Overholtz 1999, 2000). The parameter • is a shape function is almost always set to 1. To estimate per capita consumption, the gastric evacuation rate method was used (Eggers 1977, Elliott and Persson 1978). The two main parameters, • and •, were set to 0.004 and 0.11 respectively based upon prior studies and sensitivity analyses (NEFSC 2007c, 2007d). From 1992 on (when individual weights were measured), a diagnostic of % daily ration was also calculated.

Once per capita consumption rates were estimated for each monkfish size class, temporal and spatial scheme, those estimates were then scaled up to an annual and stock wide basis, *C*:

$$C = 365 \cdot C_i \cdot N_i$$

where N_i is the estimate of abundance (from assessment results) for each monkfish size class, temporal and spatial scheme and 365 is the number of days in a year.

This total consumption was partitioned for the major prey items of monkfish by multiplying it by the diet composition of each prey (D_{ij}) to provide an estimate of prey removals. Both the total consumption and the amount of prey removed by each monkfish size class (and combined across sizes) are presented as metric tons year^{-1}. These were then summed for both areas.

To evaluate the consumptive demands of a monkfish and the predatory removals of monkfish in a broader ecosystem context, total consumption by monkfish was compared to the amount of energy flow for the entire ecosystem. The total energy flows were calculated in a recent energy budget (Link et al. 2006, 2008, 2009). Monkfish consumption is presented as a percentage of total energy flows in the ecosystem. In addition, the total amount of commercially targeted prey eaten by monkfish was compared to fishery landings to evaluate potential competition between monkfish and fisheries.

Results & Observations

- The amount of food consumed by monkfish was 0.005-0.02% of all energy flows in the system
- Monkfish comprised 2-6% of total consumption by all finfish in the ecosystem (1-4 % in N, 2-8% in S)
- Consumption by monkfish has changed over time, mainly as a function of abundance (Figure A90)
- Consumption has been more important at times, perhaps when other piscivore species were at lower abundances; monkfish has the potential to be one of the dominant piscivores in the ecosystem
- All diagnostics were within the normal range.

Summary

- Amount of food eaten and per capita consumption peaked in early 1980s in both management areas; this was due to the greater abundance of large monkfish in the population.
- Total, scaled consumption follows the peak in 1980s for both management areas and early 2000s for the northern stock
- Some subtle shifts in diet across size classes, decades and areas were observed, but this species is categorically piscivorous and is of the more notable piscivores in the ecosystem
- Monkfish is an ecologically important piscivore in the Northwest Atlantic ecosystem
- Lots of small, other fishes eaten by monkfish
- Monkfish consumption (C) was high relative to landings of some of its prey stocks (L):

- C ~ 20-50% of L: mackerel, herring, monkfish
- C ~ L: squids
- C > L: silver hake, skates

TOR 8. Develop and apply analytical approaches and data that can be used for conducting single and multi-year stock projections and for computing candidate ABCs.

a. **Provide numerical short-term projections (through 2016). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. In carrying out projections, consider a range of assumptions to examine important sources of uncertainty in the assessment.**
b. **Comment on which projections seem most realistic, taking into consideration uncertainties in the assessment.**
c. **Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC.**

SCALE model results and AGEPRO projections were used to evaluate stock trends during 2011-2016 fishing at Fthreshold and at proposed ACTs and ABCs assuming stochastic long-term recruitment. Projections assumed that F in 2010 would equal the estimated F in 2009 from the SCALE model. Projections for the northern management area (NMA) are the most likely to be unrealistic, given the uncertainty of stock status due mainly to the relatively strong retrospective observed since 2002. The southern management area (SMA) projections are the more likely to be realistic, given the moderate retrospective pattern observed for that area. The combined area projections are intermediate with respect to the current management areas, as the relative scaling of the two populations is maintained when the areas are combined in one model. The projections indicate that the northern area is the most vulnerable to overfishing or becoming overfished during 2011-2016 if total catches approach the proposed ABC, while the southern area is the least vulnerable (Table A40 to Table A42).

SAW50 Editor's note: The SARC panel acknowledged the high degree of uncertainty in the projections due to uncertainty in the starting conditions (output from the SCALE model).

TOR 9. Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in recent SARC reviewed assessments and review panel reports. Identify new research recommendations.

SAW 34 (2002) Research Recommendations

***** indicates suggested candidates for deletion from the active Research Recommendations list.

1) Research should be continued to define stock structure, including genetic studies, reproductive behavior analyses, morphometric studies, parasite studies, elemental analyses, and studies of egg and larvae transport.
- *A genetic study is underway by a student at UMES using mtDNA. Results to date found genetic groupings but these are not spatially coherent (do not indicate stock separation).*

- A conventional tagging study ongoing by investigators at GMRI. Results to date: monkfish tagged in fall/winter in western Gulf of Maine and southern New England were later recaptured in Mid-Atlantic Bight (see Appendix A2). Future plans include tagging in other seasons and further to the south.
- A data storage tagging study underway, joint project of NOAA and GMRI. ~150 tagged monkfish released during 2009, no recaptures yet.
- An otolith elemental composition study is ongoing using otoliths collected during 2004 cooperative monkfish survey. Otoliths have been processed but further work has been stalled due to change in responsibilities of primary PI.
- Web site established to gather information on location of egg veils – launched spring 2007. http://www.nefsc.noaa.gov/read/popdy/monkfish/MonkfishEggveilReporting/
Results: very little response to date.
*2) The SARC recommends changing the overfishing definitions for monkfish. Research on yield per recruit for monkfish should examine the effect and possible causes of differential natural mortality rates by sex, methods to estimate gear selectivity, and the incorporation of discards.
- OF definition was changed in 2003 via Framework 2 based on results of SAW 34 and again in 2008 based on the results of NEFSC (2007a).
- NEFSC (2007a) assessment explored length-based and age-based YPR with estimates of gear selectivity from SCALE model, incorporated discards, and examined higher M to reflect shorter longevity of males. NEFSC (2007a) accepted age-based model with M=0.3, which was used to revise reference points.
*3) Surplus production modeling should continue with special emphasis placed on uncertainty in under-reported catches and population size prior to 1980.
- Bayesian surplus production was explored unsuccessfully for SAW 40 (2005) and NEFSC (2007a). The DPSWG concluded that long-term production models were inappropriate for status determination of monkfish because of the general lack of correspondence between reported catch and survey trends.
*4) Size selectivity studies should be conducted in the trawl fishery to investigate the potential effectiveness of minimum mesh size and shape regulations to reduce discards of undersize monkfish. Additionally, comparative studies of the size selectivity and catchability of trawls and gill nets should be undertaken in order to understand the differences in the numbers of large fish captured in the two gear types.
- A study using 12" diamond and square mesh was completed in 2006 (Raymond and Glass 2006). The study showed reduced catch rates of groundfish in the experimental nets compared to controls (6-6.5" mesh) and reduced discard of monkfish in the experimental nets. Monkfish was 35% of the catch (kg) in control nets and 73% in experimental nets. Discard of monkfish was reduced from 15% to 6%.
*5) Another cooperative survey for monkfish should be conducted in 2004.
- Additional cooperative surveys were conducted during 2004 and 2009. The new NEFSC survey gear is much more effective for monkfish than the previous survey gear, thus reducing the need for further cooperative surveys.
*6) Improved sampling rates (as observed in 2000-2001) for commercial landings should be maintained, which should eventually lead to an age-based assessment approach for this species.
- age sampling rates have been variable.
Observer sampling was considered more useful for monkfish by NEFSC (2007a).

NEFSC (2007a) raised concerns over the validity of ageing methods for monkfish.
7) Tagging studies should be considered as a basis to evaluate adult movement and rates of growth.
- conventional tagging study ongoing by investigators at GMRI. Results to date: monkfish tagged in fall/winter in western Gulf of Maine and southern New England were later recaptured in Mid-Atlantic Bight (see Appendix A2). Future plans include tagging in other seasons and further to the south.
- estimates of growth from conventional tagging study to date are too imprecise to estimate growth rate accurately.
- Data storage tagging study underway, joint project of NOAA and GMRI. ~150 tagged monkfish released during 2009, no recaptures yet. Fish are being marked with OTC when released for age validation studies (reward is for return of entire fish plus tags).
8) Spatial distribution of mature and immature fish and the potential effects of size limits on fishing behavior should be evaluated as a basis for advising on strategies to minimize catch and discard of immature fish.
- not done
9) Indices of abundance should be developed from industry "study fleets," including coverage from outside the depth and spatial range of the NEFSC research surveys.
- not addressed

SAW 40 Research Recommendations
*(1) An examination of the influence of fixed stations on the estimate of biomass from the cooperative research survey should be undertaken.
- As part of the 2006 cooperative monkfish survey review, catch rates, average monkfish size and density were compared between industry stations and random stations. Inclusion of the industry stations was judged to have had minimal impact on the population estimates.
*(2) An exploration of a geostatistical approach to estimate biomass from the cooperative survey would also be of value.
- not done
(3) There are some concerns with the ageing results. An ageing validation study should be undertaken to confirm the accuracy of catch at age estimates.
- Direct validation studies (e.g. tetracycline marking) have begun as part of a data storage tagging study, but no recaptures to date.
- SMAST UMass Dartmouth student working on age validation, developing tank studies (but difficult due to high mortality of captive monkfish).
- Indirect criteria have been satisfied (Armstrong et al. 1992)
*(4) The changes in the distribution in the fishery over time may be influencing the results of the assessment. This should be examined more thoroughly.
- this has not been addressed.
*(5) The assessment lacks a reliable forecast. Since commercial catch-at-age data and survey catch-at-age data exist and assuming that ageing can be validated, alternative forward-projecting age structured models should be investigated.
- a forward projecting length-tuned model (SCALE) was used to provide forecasts in the 2007 assessment and in the current assessment..
*(6) An examination of transect survey data for changes in the distribution of the population by depth would be informative.

- not done

(7) Further, consideration should be given to a more complete treatment of the Canadian portion of this stock, with possibly some interaction with the team doing the assessment of monkfish in NAFO Divisions 4VWX5Zc, possibly through the TRAC process.

- not done. There is no longer a Canadian assessment scientist assigned to monkfish; however, we have estimated survey indices from Canadian surveys on the Scotian shelf, but not incorporated them into the model.

*(8) Ways of estimating of fishing mortality at age should be investigated. This could take the form of a general linear modeling approach with survey age and year effects in an analysis of Z. Alternatively a more fully specified population model based on survey-at-age data such as the RCRV1A model of Cook (1997) and recent developments described under SURBA may be applicable.

- SCALE model is being used to estimate mortality. Survey ages alone are too variable to reliably estimate Z due to low monkfish catch rates in surveys up through 2008. With the development of a time series on the FSV Bigelow, this approach may become viable in the future.

*(9) The cooperative survey should be continued as it is informative and can be used in the Bayesian surplus production model and may provide a means of calibrating the NEFSC survey data when the survey vessel is replaced.

- A cooperative survey was conducted in 2009. Results of the 2001 and 2004 surveys were used in the surplus production models, but the modeling approach still was not successful (see SAW 34, recommendation 3). The current assessment compares the 2009 cooperative survey with the NEFSC 2009 spring survey.

2007 Data-Poor Workshop, Research Recommendations

Working Group I

(1) Observer samples should be aged.

- No further ageing has been done since NEFSC (2007a) due to questions raised about the validity of the current ageing method and because a length-based model for was adopted for the assessment.

(2) Applications of the SCALE model for monkfish assessment should be developed further, including:

*a) Explore alternative growth functions (sigmoid etc.) since von Bertalanffy growth does not fit length-at-age data

- SCALE used mean length at age, not a growth function. At present, the only growth model that would be appropriate is a linear one.

*b) Explore changing weighting on catch in relation to reliability of catch data (more uncertainty in early part of time series)

-SCALE is not currently configured to be able to do this.

*c) Explore using the same M for males and females up to age 7, and then increasing M for males to account for the lack of males over age 7

-SCALE is not currently configured to be able to do this.

*d) Bin lengths into 2cm or 5 cm increments in order to eliminate zeros in survey length frequencies

-SCALE is not currently configured to be able to do this.

e) Develop independent estimates of selectivity for application to SCALE

-No new work has been done.
*(3) Length-based mortality:
-Examine effects of vonBertalanffy growth assumption on Gedamke-Hoenig mortality estimates.
- *not done, this method was not pursued because of the adoption of the SCALE model.*
Working Group II
*(1) Investigate foreign landings and reporting rates if possible.
- *not done, not clear what is being asked for here.*
(2) Examine aging further and develop tagging studies to validate M, growth rates and Longevity
- *studies are in progress, as described above*
(3) Estimate biomass by sex since age 6+ fish that are predominantly female appear to be decreasing in biomass at a greater rate
- *not done, but could be feasible as FSV Bigelow time series accumulates*
(4) SCALE model:
a) develop objective methods for weighting input series (e.g. inverse variance weighting)
- *not done*
b) do some runs with combined management areas
- *done for current assessment*
c) develop a two-sex model
- *explored in NEFSC (2007a), but problematic because males still remain in model after none are observed in reality*
d) incorporate cannibalism in SCALE model
- *not done*
(5) examine commercial sampling length modes in more detailed time steps (e.g. quarterly) to see if cohorts can be tracked (to indicate whether there are significant problems with aging).
- *not done.*

SAW 50 Southern Demersal Working Group Research Recommendations

1. Conduct a net efficiency experiment on the FSV Bigelow to help parameterize the population models for a range of species, including monkfish.

   SAW50 Editor's note: The SARC50 panel did not comment on the Research Recommendations.

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Tables

Table A1. Timeline of events influencing fishery management of monkfish.

Month/Year	Regulatory Action
Nov. 1999	FMP implemented - Included a multi-level limited access program; two management areas; target TACs; effort limitations (DAS); Year 3 default measures (0 DAS); trip limits for limited access vessels; bycatch allowances; minimum fish sizes; minimum mesh sizes; gear restrictions; spawning season restrictions; a framework adjustment process; annual review requirements; permitting and reporting requirements; and other measures for administration and enforcement.
Nov. 1999	Amendment 1 effective – EFH Omnibus Amendment
May. 2000	DAS implemented
Jul. 2000	SAW 31
Spring 2001	Cooperative Survey
Fall 2001	Hall v. Evans decision - trip limit on gillnet vessels set equal to trawls, based on permit category.
Jan. 2002	SAW 34
Spring 2002	Councils submit Framework 1 – Proposes to fix landings at existing levels and postpone default measures for 1 year while Councils develop Amendment 2.
May. 2002	Emergency Rule – Framework 1 disapproved for non-compliance with Fthreshold in the original plan (which was invalidated by SAW 31 and SAW 34). Implemented a revision to the OFD based on SAW 34 recommendations, and management measures in FW 1.
May. 2003	Framework 2 - Modified the OFD reference points recommended by SAW 34; established an index- and landings-based method for setting TACs to achieve annual rebuilding goals; contained a method for calculating DAS and trip limits; and eliminated the default measures.
Spring 2004	Cooperative Survey
May. 2005	Amendment 2 - Made minimum fish size in SFMA equivalent to that in NFMA (11-inch tail/17-inch whole); established a 6-inch roller gear restriction in the SFMA, implemented two canyon closure areas; removed the 20-day spawning block requirement; established a research set-aside program; established an Offshore Fishery Program in the SFMA; modified some incidental catch limits; and modified the monkfish limited entry program to include vessels that had historically fished off of VA and NC.
Spring 2007	Councils submit Framework 4 - Would establish target TACs, trip limits, and DAS requirements for final 3 years of rebuilding plan; would require use of DAS in NFMA; contains backstop measures if target TACs exceeded; would revise incidental catch limits for NFMA and scallop access areas; and would adjust boundary line applicable to Category H vessels.
May. 2007	Interim Rule - Tempoarily implemented target TAC, DAS, and trip limits recommended in Framework 4 for the NFMA (except does not include the at-sea declaration provision); continues FY 2006 target TAC, DAS, and trip limits for the SFMA; and prohibits the use of carryover DAS. Also temporarily implements other measures contained in Framework 4: Revision to border applicable to Category H vessels and revisions to incidental catch limits in NFMA and scallop access areas.
Autumn 2007	Framework 4 implemented.
Apr. 2008	Framework 5 - Adopted DPWG (2007) reference point definitions, tightened loopholes (e.g. reduced DAS carryover days allowed, tightened effort accounting methods)
Oct. 2008	Framework 6 - removed backstop provision of Framework 4.
2009-2010	Amendment 5 under development to implement ACLs and AMs, and set specifications of DAS, trip limits and other management measures to replace those adopted in Framework 4. Expected to be implemented May 2011.

Table A2. Management measures for monkfish 2000-2010 (note that regulations pertain to 'fishing years,' and do not correspond to the calendar year landings in Table A3). "NMA" and "SMA": Northern and Southern Management Areas.

Target TACs, trip limits, DAS restrictions, and landings (FY 2000 - FY 2010) for NMA

		Trip Limits*	Trip Limits*			
Fishing Year	**Target TAC (mt)**	**Cat. A & C**	**Cat. B & D**	**DAS Restrictions****	**Landings (mt)**	**Percent of TAC**
2000	5,673	n/a	n/a	40	11,859	209%
2001	5,673	n/a	n/a	40	14,853	262%
2002	11,674	n/a	n/a	40	14,491	124%
2003	17,708	n/a	n/a	40	14,155	80%
2004	16,968	n/a	n/a	40	11,750	69%
2005	13,160	n/a	n/a	40	9,533	72%
2006	7,737	n/a	n/a	40	6,677	86%
2007	5,000	1,250	470	31	5,050	101%
2008	5,000	1,250	470	31	3,528	71%
2009	5,000	1,250	470	31		
2010	5,000	1,250	470	31		

* Trip limits in pounds tail weight per DAS

** Excluding up to 10 DAS carryover, became 4 DAS carryover in FY2008

Target TACs, trip limits, DAS restrictions, and landings (FY 2000 - FY 2010) for SMA

		Trip Limits*	Trip Limits*			
Fishing Year	**Target TAC (mt)**	**Cat. A & C**	**Cat. B, D, & H**	**DAS Restrictions****	**Landings (mt)**	**Percent of TAC**
2000	6,024	1,500	1,000	40	7,960	132%
2001	6,024	1,500	1,000	40	11,069	184%
2002	7,921	550	450	40	7,478	94%
2003	10,211	1,250	1,000	40	12,198	119%
2004	6,772	550	450	28	6,223	92%
2005	9,673	700	600	39.3	9,656	100%
2006	3,667	550	450	12	5,909	161%
2007	5,100	550	450	23	7,180	141%
2008	5,100	550	450	23	6,751	132%
2009	5,100	550	450	23		
2010	5,100	550	450	23		

* Trip limits in pounds tail weight per DAS

** Excluding up to 10 DAS carryover, became 4 DAS carryover in FY2008

Table A3. Landings (calculated live weight, mt) of goosefish as reported in NEFSC weighout database (1964-1993) and vessel trip reports (1994-2009) (North = SA 511-523, 561; South = SA 524-639 excluding 551-561 plus landings from North Carolina for years 1977-1995); General Canvas database (1964-1989, North = ME, NH northern weighout proportion of MA; South = Southern weighout proportion of MA, RI-VA); Foreign landings from NAFO database areas 5 and 6. Shaded cells denote suggested source for landings which are used in the total column at the far right (see text for details).

	Weigh Out Plus NC			General Canvas				
Year	US North	US South	US Total	US North	US South	US Total	Foreign	Total
1964	45	19	64	45	61	106	0	106
1965	37	17	54	37	79	115	0	115
1966	299	13	312	299	69	368	2,397	2,765
1967	539	8	547	540	59	598	11	609
1968	451	2	453	449	36	485	2,231	2,716
1969	258	4	262	240	43	283	2,249	2,532
1970	199	12	211	199	53	251	477	728
1971	213	10	223	213	53	266	3,659	3,925
1972	437	24	461	437	65	502	4,102	4,604
1973	710	139	848	708	240	948	6,818	7,766
1974	1,197	101	1,297	1,200	183	1,383	727	2,110
1975	1,853	282	2,134	1,877	417	2,294	2,548	4,842
1976	2,236	428	2,663	2,256	608	2,865	341	3,206
1977	3,137	830	3,967	3,167	1,314	4,481	275	4,756
1978	3,889	1,384	5,273	3,976	2,073	6,049	38	6,087
1979	4,014	3,534	7,548	4,068	4,697	8,765	70	8,835
1980	3,695	4,232	7,927	3,623	6,035	9,658	132	9,790
1981	3,217	2,380	5,597	3,171	4,142	7,313	381	7,694
1982	3,860	3,722	7,582	3,757	4,492	8,249	310	7,892
1983	3,849	4,115	7,964	3,918	4,707	8,624	80	8,044
1984	4,202	3,699	7,901	4,220	4,171	8,391	395	8,296
1985	4,616	4,262	8,878	4,452	4,806	9,258	1,333	10,211
1986	4,327	4,037	8,364	4,322	4,264	8,586	341	8,705
1987	4,960	3,762	8,722	4,995	3,933	8,926	748	9,470
1988	5,066	4,595	9,661	5,033	4,775	9,809	909	10,570
1989	6,391	8,353	14,744	6,263	8,678	14,910	1,178	15,922
1990	5,802	7,204	13,006				1,557	14,563
1991	5,693	9,865	15,558				1,020	16,578
1992	6,923	13,942	20,865				473	21,338
1993	10,645	15,098	25,743				354	26,097
1994	10,950	12,126	23,076				543	23,619
1995	11,970	14,361	26,331				418	27,075
1996	10,791	15,715	26,507				184	26,978
1997	9,709	18,462	28,172				189	28,517
1998	7,281	19,337	26,618				190	26,866
1999	9,128	16,085	25,213				151	25,364
2000	10,729	10,147	20,876				176	21,052
2001	13,341	9,959	23,301				142	23,450
2002	14,011	8,884	22,896				294	23,189
2003	14,991	11,095	26,086				309	26,375
2004	13,209	7,978	21,186				166	21,352
2005	10,267	8,834	19,102				206	19,308
2006	6,672	7,906	14,578				279	14,857
2007	4,855	7,290	12,145				8	12,153
2008	4,013	6,940	10,953				2	10,955
2009	3,255	5,302	8,557					8,557

Table A4. U.S. landings of monkfish (calculated live weight, mt) by gear type.

	North					South					Regions Combined				
Year	Trawl	Gill Net	Scallop Dredge	Other	Total	Trawl	Gill Net	Scallop Dredge	Other	Total	Trawl	Gill Net	Scallop Dredge	Other	Total
1964	45	0			45	19				19	64	0			64
1965	36	0			37	17				17	53	0			53
1966	299	0		0	299	13			0	13	311	0		0	312
1967	532		8		539	8				8	540		8		547
1968	447		4		451	2				2	449		4		453
1969	253	1	4		258	4				4	257	1	4		262
1970	198	0		0	199	12				12	210	0		0	211
1971	213		0		213	10				10	223		0		223
1972	426	8	1	2	437	24				24	451	8	1	2	461
1973	661	29	12	8	710	132		5	1	137	794	29	17	9	848
1974	1,060	105	7	25	1,197	98			0	98	1,160	105	7	25	1,297
1975	1,712	123	10	9	1,853	265	0	2	2	269	1,990	123	12	10	2,135
1976	2,031	143	47	15	2,236	333		7	0	340	2,459	143	54	15	2,670
1977	2,737	230	142	28	3,137	508		57	26	591	3,487	230	202	53	3,973
1978	3,255	368	212	54	3,889	605	0	507	26	1,138	4,016	368	774	80	5,238
1979	2,967	393	584	71	4,014	944	6	1,015	16	1,981	3,989	399	2,070	87	6,545
1980	2,526	518	596	56	3,696	1,139	10	1,274	7	2,429	3,723	528	2,276	62	6,589
1981	2,266	461	443	47	3,217	1,100	16	782	105	2,003	3,483	477	1,399	152	5,512
1982	3,040	421	367	32	3,860	1,806	12	1,507	27	3,352	4,998	433	2,061	60	7,551
1983	3,233	314	266	37	3,849	1,819	11	2,119	17	3,966	5,166	325	2,431	56	7,977
1984	3,648	315	196	43	4,202	1,714	15	1,704	18	3,452	5,513	330	1,968	61	7,871
1985	3,982	315	264	55	4,616	1,739	17	2,347	3	4,106	5,757	332	2,611	58	8,758
1986	3,412	326	553	36	4,327	1,841	32	2,068	12	3,954	5,318	358	2,621	48	8,345
1987	3,853	374	695	38	4,960	1,680	26	1,997	3	3,707	5,561	400	2,692	41	8,694
1988	3,554	304	1,172	36	5,066	1,828	58	2,594	3	4,483	5,399	363	3,765	39	9,567
1989	3,429	349	2,584	30	6,391	3,240	17	5,036	3	8,297	6,679	366	7,620	33	14,698
1990	3,298	338	2,141	25	5,802	2,361	32	4,744	5	7,142	5,697	372	6,885	30	12,984
1991	3,299	338	2,033	24	5,694	5,515	363	3,907	16	9,800	8,847	700	5,941	39	15,528
1992	4,330	359	2,211	24	6,923	6,528	977	6,409	11	13,925	10,860	1,336	8,619	35	20,850
1993	5,890	695	4,034	26	10,645	5,987	1,722	7,158	192	15,059	11,879	2,417	11,192	218	25,707
1994	7,574	1,571	1,808	86	11,039	5,233	2,342	3,995	556	12,126	12,707	3,884	5,759	638	22,988
1995	9,119	1,531	1,266	54	11,970	5,785	3,800	4,030	746	14,361	14,905	5,331	5,296	800	26,331
1996	8,445	1,389	913	45	10,791	7,141	4,211	4,330	33	15,715	15,586	5,599	5,243	78	26,507
1997	7,363	988	1,318	40	9,709	8,161	5,203	4,890	208	18,462	15,524	6,192	6,208	249	28,172
1998	5,421	885	948	27	7,281	7,815	6,198	5,190	134	19,337	13,236	7,083	6,138	161	26,618
1999	7,037	1,470	598	24	9,128	6,364	6,187	3,481	54	16,085	13,401	7,656	4,079	78	25,213
2000	8,234	2,102	316	76	10,729	4,018	4,005	1,975	150	10,147	12,252	6,107	2,291	226	20,876
2001	9,990	2,959	381	11	13,341	3,091	5,119	1,719	30	9,959	13,081	8,078	2,100	41	23,301
2002	10,839	2,978	181	13	14,011	1,584	5,410	1,847	43	8,884	12,423	8,389	2,028	56	22,896
2003	12,028	2,488	222	254	14,991	2,034	7,262	1,717	83	11,095	14,062	9,750	1,939	336	26,086
2004	9,918	2,866	14	411	13,209	1,228	4,605	671	1,474	7,978	11,145	7,471	685	1,885	21,186
2005	6,826	2,425	26	990	10,267	1,697	4,532	449	2,156	8,834	8,524	6,957	475	3,146	19,102
2006	4,997	1,434	33	208	6,672	1,458	3,832	377	2,238	7,906	6,455	5,265	411	2,446	14,578
2007	3,474	1,071	108	202	4,855	1,066	3,734	484	2,007	7,290	4,540	4,805	591	2,209	12,145
2008	3,048	755	19	191	4,013	1,002	3,949	360	1,629	6,940	4,050	4,705	379	1,820	10,954
2009	2,513	646	12	83	3,255	702	2,967	305	1,327	5,302	3,216	3,613	318	1,410	8,557

Table A5. Landed weight (mt) of monkfish by market category for 1964-2009 for combined assessment areas (SA 511-636), NEFSC weighout database and vessel trip reports (1994-2009).

Year	Belly Flaps	Cheeks	Livers	Gutted	Round	Dressed	Tails Unc.	Tails Large	Tails Small	Tails Peewee	All Tails
1964	0.0	0.0	0.0	0.0	0.0	0.0	19.3	0.0	0.0	0.0	19.3
1965	0.0	0.0	0.0	0.0	0.0	0.0	16.1	0.0	0.0	0.0	16.1
1966	0.0	0.0	0.0	0.0	0.0	0.0	93.9	0.0	0.0	0.0	93.0
1967	0.0	0.0	0.0	0.0	0.0	0.0	164.8	0.0	0.0	0.0	164.8
1968	0.0	0.0	0.0	0.0	0.0	0.0	136.6	0.0	0.0	0.0	136.6
1969	0.0	0.0	0.0	0.0	0.0	0.0	79.1	0.0	0.0	0.0	79.1
1970	0.0	0.0	0.0	0.0	0.0	0.0	63.5	0.0	0.0	0.0	63.5
1971	0.0	0.0	0.0	0.0	0.0	0.0	67.1	0.0	0.0	0.0	67.1
1972	0.0	0.0	0.0	0.0	0.0	0.0	139.0	0.0	0.0	0.0	139.0
1973	0.0	0.0	0.0	0.0	0.0	0.0	255.5	0.0	0.0	0.0	255.5
1974	0.0	0.0	0.0	0.0	0.0	0.0	390.7	0.0	0.0	0.0	390.7
1975	0.0	0.0	0.0	0.0	0.0	0.0	642.8	0.0	0.0	0.0	642.8
1976	0.0	0.0	0.0	0.0	0.0	0.0	802.2	0.0	0.0	0.0	802.2
1977	0.0	0.0	0.0	0.0	0.0	0.0	1194.4	0.0	0.0	0.0	1194.4
1978	0.0	0.0	0.0	0.0	0.0	0.0	1574.5	0.0	0.0	0.0	1574.5
1979	0.0	0.0	0.0	0.0	0.0	0.0	2224.7	0.0	0.0	0.0	2224.7
1980	0.0	0.0	0.0	0.0	0.0	0.0	2302.4	0.0	0.0	0.0	2302.4
1981	0.0	0.0	0.0	0.0	0.0	0.0	1654.2	0.0	0.0	0.0	1654.2
1982	0.0	0.0	10.2	0.0	0.0	0.0	2059.8	153.1	53.3	0.0	2266.2
1983	0.0	0.0	11.6	0.0	0.0	0.0	2009.9	241.4	138.6	0.0	2390.0
1984	0.0	0.0	25.0	0.0	0.0	0.0	2121.6	186.8	44.5	0.0	2352.9
1985	0.0	0.0	28.0	0.0	0.0	0.0	2467.0	86.7	73.4	0.0	2627.1
1986	0.0	0.0	36.3	0.0	0.0	0.0	2365.4	76.4	52.2	0.0	2494.0
1987	0.0	0.0	54.2	0.0	0.0	0.0	2463.7	139.9	6.7	0.0	2610.3
1988	0.0	0.0	112.8	0.0	0.0	0.0	2646.3	195.1	34.8	0.0	2876.2
1989	0.0	0.0	146.3	0.0	15.6	0.0	3501.8	557.4	360.0	0.0	4419.2
1990	0.0	0.0	179.7	0.0	217.7	0.0	2601.8	854.1	377.4	0.0	3833.3
1991	0.0	8.6	270.3	0.0	415.4	0.0	2229.1	1661.9	614.1	36.6	4541.6
1992	0.2	3.7	321.5	0.0	386.0	0.0	2778.7	1908.1	1293.0	183.3	6163.1
1993	0.0	1.7	459.9	98.2	528.7	0.0	3503.2	1933.0	1851.1	262.4	7549.8
1994	0.0	5.3	458.1	1453.6	2044.8	0.0	1256.9	2230.7	2063.3	258.0	5808.9
1995	2.3	1.0	497.0	2752.4	2652.4	0.0	879.7	2521.4	2422.6	363.3	6187.1
1996	0.4	0.6	569.5	3467.8	1063.1	0.0	1086.0	2090.1	3027.2	269.6	6472.9
1997	0.1	0.1	628.0	3193.7	795.2	0.0	673.6	3050.1	3274.0	151.5	7149.3
1998	0.0	0.5	605.9	3586.9	581.8	0.0	858.3	3006.8	2649.8	95.5	6610.4
1999	0.1	0.2	597.4	5748.1	1131.4	0.0	537.2	2388.3	2200.8	153.4	5279.8
2000	0.0	3.7	624.0	6914.1	1091.0	0.0	293.6	1580.0	1707.3	4.3	3585.1
2001	0.5	0.0	559.4	7028.2	531.4	0.0	345.3	1958.9	2140.3	0.4	4444.9
2002	0.2	0.1	508.7	7801.7	575.4	0.0	246.6	1683.9	2113.3	0.2	4044.0
2003	0.0	1.0	486.3	7322.8	680.9	0.0	337.1	2362.6	2437.4	0.7	5137.8
2004	0.3	2.1	410.7	3404.6	2026.0	7.8	188.6	2553.4	1853.9	1.5	4597.4
2005	0.0	54.9	373.5	3361.0	2334.3	17.7	107.4	2209.9	1564.7	3.7	3885.6
2006	0.1	108.4	312.1	2972.8	2002.0	21.4	77.4	1548.2	1125.8	3.3	2754.7
2007	0.0	43.7	271.2	2340.1	1478.2	12.3	96.5	1596.5	707.3	1.8	2402.0
2008	0.0	4.8	256.8	2138.9	1280.5	15.4	60.1	1502.5	607.1	0.0	2169.8
2009	0.8	0.0	199.1	1692.9	1119.5	19.4	47.8	1065.0	534.0	0.3	1647.1

Table A6. Landed weight (mt) of monkfish by market category for 1964-2009 for northern assessment area (SA 511-523 and 561), NEFSC weighout database and vessel trip reports (1994-2009).

Year	Belly Flaps	Cheeks	Livers	Gutted	Round	Dressed	Heads	Tails Unc.	Tails Large	Tails Small	Tails Peewee	All Tails
1964	0.0	0.0	0.0	0.0	0.0	0.0	0.0	13.5	0.0	0.0	0.0	13.5
1965	0.0	0.0	0.0	0.0	0.0	0.0	0.0	11.0	0.0	0.0	0.0	11.0
1966	0.0	0.0	0.0	0.0	0.0	0.0	0.0	90.1	0.0	0.0	0.0	90.1
1967	0.0	0.0	0.0	0.0	0.0	0.0	0.0	162.5	0.0	0.0	0.0	162.5
1968	0.0	0.0	0.0	0.0	0.0	0.0	0.0	135.9	0.0	0.0	0.0	135.9
1969	0.0	0.0	0.0	0.0	0.0	0.0	0.0	77.8	0.0	0.0	0.0	77.8
1970	0.0	0.0	0.0	0.0	0.0	0.0	0.0	59.8	0.0	0.0	0.0	59.8
1971	0.0	0.0	0.0	0.0	0.0	0.0	0.0	64.1	0.0	0.0	0.0	64.1
1972	0.0	0.0	0.0	0.0	0.0	0.0	0.0	131.6	0.0	0.0	0.0	131.6
1973	0.0	0.0	0.0	0.0	0.0	0.0	0.0	213.8	0.0	0.0	0.0	213.8
1974	0.0	0.0	0.0	0.0	0.0	0.0	0.0	360.4	0.0	0.0	0.0	360.4
1975	0.0	0.0	0.0	0.0	0.0	0.0	0.0	558.0	0.0	0.0	0.0	558.0
1976	0.0	0.0	0.0	0.0	0.0	0.0	0.0	673.4	0.0	0.0	0.0	673.4
1977	0.0	0.0	0.0	0.0	0.0	0.0	0.0	944.7	0.0	0.0	0.0	944.7
1978	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1171.4	0.0	0.0	0.0	1171.4
1979	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1209.1	0.0	0.0	0.0	1209.1
1980	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1113.1	0.0	0.0	0.0	1113.1
1981	0.0	0.0	0.0	0.0	0.0	0.0	0.0	969.0	0.0	0.0	0.0	969.0
1982	0.0	0.0	10.0	0.0	0.0	0.0	0.0	1145.6	15.0	2.0	0.0	1162.6
1983	0.0	0.0	9.3	0.0	0.0	0.0	0.0	1152.3	4.8	2.4	0.0	1159.4
1984	0.0	0.0	14.7	0.0	0.0	0.0	0.0	1261.9	3.7	0.0	0.0	1265.6
1985	0.0	0.0	11.4	0.0	0.0	0.0	0.0	1385.9	1.6	2.6	0.0	1390.2
1986	0.0	0.0	13.7	0.0	0.0	0.0	0.0	1302.7	0.3	0.2	0.0	1303.2
1987	0.0	0.0	24.0	0.0	0.0	0.0	0.0	1491.5	1.7	0.7	0.0	1493.9
1988	0.0	0.0	47.4	0.0	0.0	0.0	0.0	1516.9	5.6	3.3	0.0	1525.8
1989	0.0	0.0	58.7	0.0	11.2	0.0	0.0	1464.5	327.0	130.2	0.0	1921.6
1990	0.0	0.0	77.9	0.0	30.3	0.0	0.0	1173.7	410.7	154.0	0.0	1738.4
1991	0.0	3.3	70.0	0.0	0.3	0.0	0.0	1013.9	538.6	153.2	9.1	1714.8
1992	0.0	0.7	83.0	0.0	0.1	0.0	0.0	910.5	589.9	505.4	79.4	2085.3
1993	0.0	0.6	208.3	98.2	350.6	0.0	0.0	1034.3	867.9	1061.8	102.9	3067.0
1994	0.0	1.4	207.6	532.7	981.3	0.0	0.0	403.0	1205.7	1074.8	136.2	2819.7
1995	0.0	0.7	45.7	1223.7	1113.3	0.0	0.0	361.7	1180.4	1003.3	304.4	2849.9
1996	0.3	0.2	65.1	1115.7	745.4	0.0	0.0	89.8	930.4	1398.6	223.9	2642.7
1997	0.0	0.1	50.9	634.3	244.3	0.0	0.0	26.4	1126.1	1361.5	119.1	2633.1
1998	0.0	0.0	24.0	550.9	143.9	0.0	0.0	16.3	1054.9	810.1	79.2	1960.5
1999	0.0	0.1	39.8	1700.8	510.6	0.0	0.0	28.3	995.5	848.4	139.4	2011.6
2000	0.0	0.0	93.9	3213.4	912.1	0.0	0.0	17.5	782.9	1050.4	2.7	1853.4
2001	0.0	0.0	93.5	3084.2	231.1	0.0	0.0	128.5	1114.6	1646.7	0.0	2889.8
2002	0.0	0.1	75.3	3788.7	24.1	0.0	0.0	79.6	1055.3	1777.2	0.0	2912.0
2003	0.0	0.0	60.6	2363.9	13.7	0.0	0.0	94.7	1572.5	2032.2	0.0	3699.5
2004	0.0	0.0	55.8	646.7	959.9	0.0	0.0	3.0	1882.5	1580.3	1.4	3467.3
2005	0.0	0.0	41.2	732.9	953.0	0.1	0.0	2.3	1498.5	1051.4	1.6	2553.8
2006	0.0	0.0	22.4	865.3	715.7	1.0	0.0	7.6	881.9	604.7	2.6	1496.9
2007	0.0	0.1	13.2	299.9	319.3	0.1	0.6	8.4	868.3	385.6	0.8	1263.1
2008	0.0	0.0	4.2	203.5	160.6	2.0	0.0	1.3	780.2	307.9	0.0	1089.3
2009	0.0	0.0	2.03	116.51	189.58	10.69	0.0	1.0	573.05	302.7	0.0	876.7

Table A7. Landed weight (mt) of monkfish by market category for 1964-2009 for southern assessment area (SA 524-636 excluding 561), NEFSC weighout database and vessel trip reports (1994-2009).

Year	Belly Flaps	Cheeks	Livers	Gutted	Round	Dressed	Heads	Tails Unc.	Tails Large	Tails Small	Tails Peewee	All Tails
1964	0.0	0.0	0.0	0.0	0.0	0.0		5.7	0.0	0.0	0.0	5.7
1965	0.0	0.0	0.0	0.0	0.0	0.0		5.0	0.0	0.0	0.0	5.0
1966	0.0	0.0	0.0	0.0	0.0	0.0		3.9	0.0	0.0	0.0	3.8
1967	0.0	0.0	0.0	0.0	0.0	0.0		2.3	0.0	0.0	0.0	2.3
1968	0.0	0.0	0.0	0.0	0.0	0.0		0.6	0.0	0.0	0.0	0.6
1969	0.0	0.0	0.0	0.0	0.0	0.0		1.2	0.0	0.0	0.0	1.2
1970	0.0	0.0	0.0	0.0	0.0	0.0		3.7	0.0	0.0	0.0	3.7
1971	0.0	0.0	0.0	0.0	0.0	0.0		3.0	0.0	0.0	0.0	3.0
1972	0.0	0.0	0.0	0.0	0.0	0.0		7.4	0.0	0.0	0.0	7.4
1973	0.0	0.0	0.0	0.0	0.0	0.0		41.7	0.0	0.0	0.0	41.7
1974	0.0	0.0	0.0	0.0	0.0	0.0		30.3	0.0	0.0	0.0	30.3
1975	0.0	0.0	0.0	0.0	0.0	0.0		84.8	0.0	0.0	0.0	84.8
1976	0.0	0.0	0.0	0.0	0.0	0.0		128.8	0.0	0.0	0.0	128.8
1977	0.0	0.0	0.0	0.0	0.0	0.0		249.6	0.0	0.0	0.0	249.6
1978	0.0	0.0	0.0	0.0	0.0	0.0		403.1	0.0	0.0	0.0	403.1
1979	0.0	0.0	0.0	0.0	0.0	0.0		1015.6	0.0	0.0	0.0	1015.6
1980	0.0	0.0	0.0	0.0	0.0	0.0		1189.3	0.0	0.0	0.0	1189.3
1981	0.0	0.0	0.0	0.0	0.0	0.0		685.0	0.0	0.0	0.0	685.0
1982	0.0	0.0	0.2	0.0	0.0	0.0		912.4	138.1	51.3	0.0	1101.8
1983	0.0	0.0	2.3	0.0	0.0	0.0		857.7	236.6	136.2	0.0	1230.5
1984	0.0	0.0	10.3	0.0	0.0	0.0		859.7	183.1	44.5	0.0	1087.3
1985	0.0	0.0	16.7	0.0	0.0	0.0		1081.1	85.1	70.8	0.0	1236.9
1986	0.0	0.0	22.6	0.0	0.0	0.0		1062.6	76.1	52.0	0.0	1190.8
1987	0.0	0.0	330.2	0.0	0.0	0.0		972.2	138.2	6.0	0.0	1116.4
1988	0.0	0.0	65.4	0.0	0.0	0.0		1129.3	189.5	31.5	0.0	1350.4
1989	0.0	0.0	87.6	0.0	4.5	0.0		2037.4	230.4	229.8	0.0	2497.5
1990	0.0	0.0	101.8	0.0	187.3	0.0		1428.1	443.4	223.4	0.0	2094.9
1991	0.0	5.2	200.2	0.0	415.1	0.0		1215.2	1123.3	460.9	27.5	2826.8
1992	0.2	3.0	238.5	0.0	385.9	0.0		1868.2	1318.3	787.6	103.9	4077.9
1993	0.0	1.1	251.5	0.0	178.1	0.0		2468.9	1065.1	789.3	159.4	4482.8
1994	0.0	3.8	250.5	921.0	1063.5	0.0		853.9	1025.0	988.5	121.8	2989.2
1995	2.3	0.3	451.3	1528.7	1539.1	0.0		518.0	1341.0	1419.3	58.9	3337.2
1996	0.4	0.5	504.4	2352.1	317.6	0.0		996.3	1159.7	1628.6	45.6	3830.2
1997	0.1	0.0	577.1	2559.4	550.9	0.0		647.2	1924.0	1912.6	32.4	4516.2
1998	0.0	0.5	581.9	3036.0	438.0	0.0		841.9	1952.0	1839.7	16.3	4649.9
1999	0.1	0.1	557.6	4047.4	620.9	0.0		508.9	1392.8	1352.4	14.1	3268.1
2000	0.0	3.7	530.1	3700.7	178.9	0.0		276.2	797.1	656.9	1.6	1731.8
2001	0.5	0.0	465.9	3944.0	300.3	0.0		216.8	844.3	493.6	0.4	1555.1
2002	0.2	0.0	433.3	4012.9	551.3	0.0		167.0	628.6	336.1	0.2	1132.0
2003	0.0	0.9	425.7	4958.8	667.2	0.0		242.4	790.1	405.1	0.7	1438.3
2004	0.3	2.1	354.9	2758.0	1066.1	7.8		185.6	670.8	273.6	0.1	1130.1
2005	0.0	54.9	332.3	2628.1	1381.3	17.7		105.0	711.3	513.3	2.1	1331.8
2006	0.1	108.4	289.6	2107.5	1286.3	20.4		69.8	666.3	521.1	0.7	1257.9
2007	0.0	43.6	258.0	2040.2	1158.9	12.2	0.1	88.2	728.2	321.7	0.9	1138.9
2008	0.0	4.8	252.6	1935.4	1119.9	13.4	1.1	58.8	722.4	299.3	0.0	1080.5
2009	0.8	0.0	197.0	1576.4	929.9	8.7	11.4	46.9	491.9	231.3	0.3	770.4

Table A8. Revised discard estimates. Dredge and shrimp trawl based on SBRM d/k all species, live weight; trawl and gillnet based on revised d/k monk in the northern and southern management areas.

North							
GEAR	YEAR	HALF	No. Trips	D/K Ratio	CV	mt	Monkfish Discard (mt)
Trawl	1989	1	17	0.041	0.63	1,550	63
		2	50	0.182	0.44	1,830	333
	1990	1	9	0.089	0.71	1,589	141
		2	30	0.040	0.46	1,694	68
	1991	1	21	0.043	0.47	1,239	53
		2	53	0.210	0.19	2,027	427
	1992	1	40	0.132	0.32	1,675	222
		2	18	0.266	0.38	2,625	698
	1993	1	8	0.076	0.36	2,821	216
		2	12	0.089	0.25	3,032	270
	1994	1	5	0.040	0.46	2,899	115
		2	4	0.037	0.44	4,353	161
	1995	1	22	0.154	0.32	4,224	652
		2	45	0.088	0.32	4,630	407
	1996	1	14	0.196	0.25	4,210	827
		2	41	0.134	0.57	4,188	559
	1997	1	10	0.099	0.49	3,364	332
		2	7	0.076	0.23	3,444	260
	1998	1	6	0.112	0.37	2,736	306
		2	3	0.088	0.09	2,376	210
	1999	1	2	0.098	0.04	3,742	368
		2	27	0.070	0.22	3,226	226
	2000	1	49	0.074	0.40	4,522	334
		2	53	0.081	0.21	4,200	341
	2001	1	40	0.099	0.22	5,564	553
		2	99	0.064	0.11	5,090	326
	2002	1	28	0.078	0.31	6,235	489
		2	198	0.102	0.12	5,037	514
	2003	1	123	0.099	0.16	7,256	717
		2	169	0.052	0.13	5,340	280
	2004	1	86	0.041	0.13	5,942	242
		2	225	0.045	0.14	4,120	184
	2005	1	55	0.091	0.36	3,825	348
		2	348	0.101	0.14	2,812	285
	2006	1	93	0.041	0.15	2,837	116
		2	58	0.083	0.13	2,259	189
	2007	1	53	0.039	0.14	2,133	82
		2	100	0.083	0.21	1,467	122
	2008	1	66	0.090	0.17	1,890	170
		2	95	0.121	0.23	1,285	155
	2009	1	74	0.204	0.17	1,731	353
		2	114	0.103	0.16	837	86

North							
GEAR	YEAR	HALF	No. Trips	D/K Ratio	CV	mt	Monkfish Discard (mt)
Gillnet	1989	1	1	0.000		84	0
		2	77	0.027	0.32	265	7
	1990	1	37	0.036	0.42	121	4
		2	51	0.029	0.37	219	6
	1991	1	131	0.030	0.48	120	4
		2	555	0.036	0.11	213	8
	1992	1	216	0.065	0.17	105	7
		2	430	0.040	0.25	248	10
	1993	1	106	0.084	0.22	119	10
		2	261	0.032	0.24	560	18
	1994	1	19	0.065	0.30	132	9
		2	38	0.054	0.20	959	52
	1995	1	26	0.141	0.31	334	47
		2	67	0.087	0.23	1,242	109
	1996	1	19	0.137	0.43	348	48
		2	31	0.131	0.19	1,063	140
	1997	1	15	0.036	0.32	244	9
		2	23	0.194	0.84	867	168
	1998	1	27	0.028	0.41	196	5
		2	63	0.043	0.28	746	32
	1999	1	27	0.067	0.66	344	23
		2	59	0.036	0.51	1,088	39
	2000	1	40	0.037	0.24	500	18
		2	59	0.077	0.24	1,879	145
	2001	1	25	0.061	0.70	919	56
		2	30	0.849	0.94	2,227	1,892
	2002	1	19	0.040	0.57	821	33
		2	38	0.048	0.30	2,127	103
	2003	1	83	0.037	0.24	567	21
		2	208	0.053	0.14	1,791	94
	2004	1	91	0.022	0.25	826	19
		2	504	0.054	0.12	2,067	112
	2005	1	37	0.106	0.29	545	58
		2	523	0.071	0.10	1,567	112
	2006	1	49	0.066	0.43	357	23
		2	48	0.082	0.18	1,172	96
	2007	1	22	0.059	0.32	291	17
		2	147	0.065	0.18	847	55
	2008	1	39	0.079	0.30	183	14
		2	94	0.047	0.25	634	30
	2009	1	27	0.202	0.47	190	38
		2	90	0.076	0.21	484	37

Table A8. continued (north)

North							
GEAR	YEAR	HALF	No. Trips	D/K Ratio	CV	mt	Monkfish Discard (mt)
Shrimp	1989	1	31	0.002	0.34	3,412	6
		2	9	0.001	0.62	931	1
	1990	1	27	0.020	0.34	4,548	92
		2	4	0.020	1.01	620	13
	1991	1	46	0.020	0.19	3,536	71
		2	7	0.020	0.40	340	7
	1992	1	76	0.003	0.23	3,285	10
		2	6	0.003	0.28	161	0
	1993	1	78	0.001	0.26	1,890	2
		2	4	0.001	0.70	316	0
	1994	1	69	0.002	0.39	2,431	6
		2	6	0.001	0.44	1,118	1
	1995	1	62	0.000	0.24	5,416	2
		2	9	0.001	0.43	1,509	1
	1996	1	31	0.000	0.34	7,687	1
		2	5	0.000	0.79	1,475	0
	1997	1	17	0.000	0.61	5,659	1
		2		0.001		655	0
	1998	1		0.000		3,423	1
		2		0.001		160	0
	1999	1		0.000		1,578	0
		2					
	2000	1		0.000		2,238	1
		2		0.001		98	0
	2001	1	3	0.000	0.14	1,094	0
		2					
	2002	1		0.000		417	0
		2					
	2003	1	13	0.000	1.00	1,017	0
		2					
	2004	1	12	0.000	0.25	1,518	0
		2		0.001		24	0
	2005	1	16	0.000	0.53	830	0
		2		0.001		56	0
	2006	1	10	0.000	0.72	618	0
		2	3	0.000	0.10	189	0
	2007	1	9	0.001	0.89	1,600	1
		2	0	0.000	0.00	217	0
	2008	1	15	0.000	1.04	1,763	1
		2	3	0.001	0.90	50	0
	2009	1	7	0.001	0.62	433	0
		2	0	0.000	0.00	25	0

North							
GEAR	YEAR	HALF	No. Trips	D/K Ratio	CV	mt	Monkfish Discard (mt)
Dredge	1989	1		0.002		18,213	37
		2		0.020		24,053	485
	1990	1		0.002		9,864	20
		2		0.020		19,293	389
	1991	1		0.002		16,608	34
		2		0.020		21,313	430
	1992	1		0.002		14,179	29
		2	1	0.003		20,033	56
	1993	1	2	0.002	0.05	13,702	27
		2	2	0.027	0.24	12,665	341
	1994	1	1	0.003		5,477	15
		2	2	0.006	0.64	4,500	27
	1995	1		0.002		2,915	6
		2	1	0.036		8,435	305
	1996	1	4	0.000	0.63	12,015	3
		2	1	0.034		12,182	420
	1997	1	3	0.004	0.79	19,009	69
		2	3	0.025	0.87	19,866	502
	1998	1	1	0.004		20,980	89
		2	2	0.017	0.07	16,979	281
	1999	1	1	0.002		27,495	65
		2		0.002		29,283	69
	2000	1		0.004		29,383	120
		2	84	0.004	0.15	13,809	56
	2001	1	13	0.003	0.52	16,174	44
		2		0.003		12,512	34
	2002	1		0.015		9,478	138
		2	5	0.015	0.95	11,713	170
	2003	1	3	0.000	1.50	17,082	2
		2	2	0.019	0.74	10,855	204
	2004	1	2	0.000		4,269	0
		2	7	0.276	0.61	1,080	298
	2005	1	15	0.001	0.60	2,427	3
		2	29	0.007	0.24	11,761	87
	2006	1	2	0.000	0.81	8,869	4
		2	10	0.010	0.36	5,445	54
	2007	1	19	0.002	0.22	3,096	6
		2	42	0.022	0.22	6,309	137
	2008	1	8	0.002	0.28	1,840	3
		2	10	0.007	0.57	1,016	7
	2009	1	2	0.013	0.09	593	7
		2	12	0.002	0.25	3,418	7

Table A8. continued (south)

South							
GEAR	YEAR	HALF	No. Trips	D/K Ratio	CV	mt	Monkfish Discard (mt)
Trawl	1989	1	37	0.791	0.37	2,195	1,736
		2	29	0.175	0.55	733	128
	1990	1	36	0.063	0.25	1,540	98
		2	19	0.114	0.33	755	86
	1991	1	51	0.255	0.30	1,251	319
		2	59	0.020	0.38	3,804	78
	1992	1	54	0.059	0.37	3,946	232
		2	25	0.028	0.84	2,134	60
	1993	1	36	0.089	0.59	2,598	232
		2	23	0.027	0.50	1,301	35
	1994	1	35	0.068	0.29	3,039	205
		2	18	0.228	0.63	2,089	477
	1995	1	43	0.150	0.41	3,252	488
		2	31	0.113	0.49	2,709	307
	1996	1	42	0.156	0.30	3,154	491
		2	29	0.094	0.19	3,818	359
	1997	1	43	0.025	0.47	4,355	107
		2	18	0.089	0.15	4,015	356
	1998	1	28	0.120	0.29	4,321	517
		2	15	0.027	0.52	3,648	100
	1999	1	29	0.050	0.36	4,180	209
		2	17	0.211	0.58	2,119	448
	2000	1	54	0.197	0.49	1,766	347
		2	37	0.102	0.52	1,645	167
	2001	1	42	1.551	0.46	1,460	2,265
		2	26	0.368	0.64	959	353
	2002	1	37	0.127	0.55	833	106
		2	30	0.128	0.25	314	40
	2003	1	94	0.156	0.24	712	111
		2	63	0.249	0.38	750	187
	2004	1	158	0.189	0.43	824	156
		2	176	0.981	0.36	755	740
	2005	1	149	0.592	0.34	730	432
		2	210	0.344	0.31	1,608	553
	2006	1	148	0.382	0.22	904	345
		2	102	0.130	0.35	925	121
	2007	1	142	0.228	0.45	660	150
		2	147	0.376	0.59	817	307
	2008	1	135	0.198	0.31	712	141
		2	94	0.062	0.44	609	38
	2009	1	115	0.085	0.33	593	51
		2	75	0.087	0.69	366	32

South							
GEAR	YEAR	HALF	No. Trips	D/K Ratio	CV	mt	Monkfish Discard (mt)
Gillnet	1989	1		0.031		12	0
		2		0.054		5	0
	1990	1		0.031		14	0
		2		0.054		18	1
	1991	1		0.031		209	7
		2	2	0.008	0.16	154	1
	1992	1	60	0.011	0.32	786	8
		2	41	0.020	0.20	176	4
	1993	1	50	0.034	0.71	1,306	44
		2	45	0.059	0.24	341	20
	1994	1	46	0.079	0.34	1,649	130
		2	61	0.058	0.19	830	48
	1995	1	156	0.038	0.19	2,810	108
		2	44	0.041	0.30	937	39
	1996	1	123	0.071	0.28	2,795	199
		2	14	0.052	0.30	1,363	70
	1997	1	150	0.070	0.35	3,688	257
		2	31	0.015	0.35	1,320	19
	1998	1	105	0.067	0.22	4,172	278
		2	13	0.063	0.46	1,948	122
	1999	1	22	0.052	0.35	4,338	227
		2	6	0.046	0.62	1,829	84
	2000	1	22	0.063	0.31	2,688	170
		2	10	0.056	0.93	1,034	58
	2001	1	16	0.030	0.44	2,175	65
		2	4	0.033	0.44	2,758	91
	2002	1	11	0.017	0.83	3,506	60
		2	7	0.063	0.47	1,933	122
	2003	1	31	0.016	0.35	4,671	73
		2	39	0.070	0.32	2,721	190
	2004	1	55	0.062	0.26	3,767	232
		2	43	0.096	0.26	1,221	118
	2005	1	66	0.127	0.23	3,586	456
		2	39	0.080	0.29	1,724	138
	2006	1	36	0.051	0.21	3,151	162
		2	7	0.087	0.37	1,034	89
	2007	1	26	0.228	0.41	2,922	666
		2	17	0.059	0.33	2,217	132
	2008	1	27	0.108	0.35	3,853	417
		2	18	0.121	0.30	1,290	156
	2009	1	29	0.054	0.25	3,035	164
		2	5	0.093	0.22	868	81

Table A8. continued (south)

South							
GEAR	YEAR	HALF	No. Trips	D/K Ratio	CV	mt	Monkfish Discard (mt)
Dredge	1989	1		0.012		59,697	706
		2		0.013		35,498	455
	1990	1		0.012		64,315	761
		2		0.013		53,041	679
	1991	1		0.012		67,830	802
		2	2	0.001	0.25	36,015	22
	1992	1	7	0.000	0.80	48,687	20
		2	7	0.006	0.62	39,127	253
	1993	1	11	0.008	0.29	23,971	184
		2	3	0.029	0.78	18,379	532
	1994	1	9	0.022	0.24	22,841	512
		2	8	0.015	0.29	27,175	420
	1995	1	14	0.029	0.17	34,832	1,016
		2	8	0.041	0.47	18,089	746
	1996	1	18	0.017	0.25	21,250	370
		2	14	0.024	0.28	18,878	448
	1997	1	16	0.026	0.21	10,175	261
		2	7	0.035	0.41	4,329	152
	1998	1	8	0.008	0.27	4,284	33
		2	15	0.011	0.55	4,700	53
	1999	1	2	0.016	0.18	11,695	192
		2	12	0.006	0.52	12,136	72
	2000	1	36	0.015	0.16	26,596	389
		2	132	0.008	0.17	42,541	360
	2001	1	44	0.014	0.12	62,987	907
		2	48	0.014	0.15	69,336	964
	2002	1	34	0.019	0.09	84,180	1,575
		2	55	0.018	0.10	81,242	1,479
	2003	1	46	0.014	0.16	82,123	1,138
		2	71	0.017	0.12	92,174	1,522
	2004	1	74	0.014	0.09	71,786	1,024
		2	164	0.014	0.10	30,188	430
	2005	1	98	0.012	0.14	41,192	500
		2	147	0.016	0.13	29,264	466
	2006	1	42	0.008	0.31	28,640	243
		2	135	0.024	0.14	35,961	846
	2007	1	130	0.010	0.14	27,584	278
		2	156	0.014	0.14	17,512	241
	2008	1	367	0.006	0.11	28,746	181
		2	241	0.010	0.14	20,230	197
	2009	1	318	0.006	0.09	36,251	213
		2	67	0.011	0.15	25,095	266

Table A9. Estimated discards of monkfish using SBRM methodology (mt monkfish discarded/mt all species landed) in trawls, gillnets, and scallop dredge

Stock	Year	Trawl	Gillnet	Scallop Dredge	Total
North	1989	119	15	465	599
	1990	183	12	321	515
	1991	357	19	417	792
	1992	444	20	56	520
	1993	186	21	368	575
	1994	237	117	56	410
	1995	1,295	148	354	1,797
	1996	1,398	156	383	1,938
	1997	730	152	302	1,184
	1998	610	30	167	807
	1999	774	34	53	861
	2000	766	214	100	1,079
	2001	1,193	1,671	80	2,944
	2002	1,069	116	321	1,507
	2003	1,090	151	215	1,455
	2004	543	101	1,079	1,723
	2005	437	194	55	686
	2006	283	74	37	394
	2007	204	73	143	420
	2008	325	44	10	380
	2009	439	75	14	528
North Total		12,683	3,436	4,996	21,115
South	1989	919	29	43	991
	1990	205	19	64	289
	1991	246	40	22	307
	1992	656	21	273	950
	1993	296	169	716	1,181
	1994	1,126	39	850	2,015
	1995	1,509	44	1,818	3,372
	1996	222	73	935	1,230
	1997	254	171	919	1,344
	1998	155	184	267	607
	1999	771	220	623	1,614
	2000	411	214	1,023	1,647
	2001	420	80	1,860	2,361
	2002	514	172	3,038	3,724
	2003	536	331	2,649	3,516
	2004	964	979	1,129	3,072
	2005	688	1,519	665	2,872
	2006	288	502	732	1,523
	2007	458	798	519	1,775
	2008	179	573	378	1,130
	2009	82	245	479	806
South Total		10,901	6,424	19,002	36,327
Grand Total		23,584	9,860	23,998	57,442

Table A10. Annual catch, discards using (mt monks discarded/mt kept of all species) for dredges and shrimp trawls and (mt monks discarded/mt monks kept) for trawls and gillnets. The new estimates also reflect minor changes to allocation to stock based on live weight rather than landed weight. Foreign is NAFO areas 5 and 6

	North			South			Areas Combined				
Year	Landings	Discard	Total (mt)	Landings	Discard	Total (mt)	Landings	Discard	Total (mt)	Foreign	Total (mt)
1980	3,623	767	4,390	6,035	395	6,430	9,658	1,163	10,821	132	10,953
1981	3,171	916	4,087	4,142	319	4,461	7,313	1,235	8,548	381	8,929
1982	3,860	841	4,701	3,722	417	4,139	7,582	1,258	8,840	310	9,150
1983	3,849	797	4,646	4,115	467	4,582	7,964	1,264	9,228	80	9,308
1984	4,202	733	4,935	3,699	483	4,182	7,901	1,216	9,117	395	9,512
1985	4,616	757	5,373	4,262	451	4,713	8,878	1,208	10,086	1,333	11,419
1986	4,327	652	4,979	4,037	439	4,476	8,364	1,091	9,455	341	9,796
1987	4,960	914	5,874	3,762	726	4,488	8,722	1,640	10,362	748	11,110
1988	5,066	942	6,008	4,595	721	5,316	9,661	1,664	11,325	909	12,234
1989	6,391	932	7,323	8,353	3,026	11,379	14,744	3,958	18,702	1,178	19,880
1990	5,802	733	6,535	7,204	1,626	8,830	13,006	2,359	15,365	1,557	16,922
1991	5,693	1,033	6,726	9,865	1,229	11,094	15,558	2,262	17,820	1,020	18,840
1992	6,923	1,031	7,954	13,942	577	14,519	20,865	1,608	22,473	473	22,946
1993	10,645	885	11,530	15,098	1,047	16,145	25,743	1,932	27,675	354	28,029
1994	10,950	385	11,335	12,126	1,793	13,919	23,076	2,178	25,254	543	25,797
1995	11,970	1,530	13,500	14,361	2,703	17,064	26,331	4,232	30,564	418	30,982
1996	10,791	1,998	12,789	15,715	1,937	17,652	26,507	3,934	30,441	184	30,625
1997	9,709	1,341	11,051	18,462	1,152	19,614	28,172	2,494	30,665	189	30,854
1998	7,281	924	8,205	19,337	1,102	20,438	26,618	2,026	28,643	190	28,833
1999	9,128	790	9,918	16,085	1,231	17,316	25,213	2,021	27,234	151	27,385
2000	10,729	1,015	11,743	10,147	1,491	11,638	20,876	2,506	23,382	176	23,558
2001	13,341	2,904	16,245	9,959	4,645	14,604	23,301	7,549	30,849	149	30,998
2002	14,011	1,446	15,457	8,884	3,382	12,266	22,896	4,828	27,724	294	28,018
2003	14,991	1,318	16,309	11,095	3,220	14,316	26,086	4,538	30,625	309	30,934
2004	13,209	854	14,062	7,978	2,699	10,677	21,186	3,553	24,739	166	24,905
2005	10,267	892	11,159	8,834	2,546	11,380	19,102	3,438	22,540	206	22,746
2006	6,706	481	7,187	7,755	1,806	9,561	14,461	2,288	16,748	279	17,027
2007	4,855	421	5,276	7,290	1,775	9,065	12,145	2,196	14,341	8	14,349
2008	4,013	380	4,393	6,940	1,130	8,070	10,953	1,510	12,463	2	12,465
2009	3,255	528	3,783	5,302	806	6,108	8,557	1,334	9,891		9,891

Table A11. Temporal stratification used in expanding landings and discard to length composition of the monkfish catch. Unless otherwise indicated, sampling was expanded within gear type and area.

		Trawl		Gillnet		Dredge	
		Kept	Discarded	Kept	Discarded	Kept	Discarded
2007	North	half year	half year	annual	annual N+S	annual	annual
	South	half year	half year	annual	annual	annual	annual
2008	North	half year	half year	annual	annual N+S	annual	annual
	South	half year	half year	annual	annual	annual	annual
2009	North	half year	half year	annual	annual N+S	annual	annual
	South	half year	half year	annual	annual	annual	annual

Table A12. Number of tows from 2009 cooperative monkfish survey with sensor data.

	MK Cookie	ER Cookie	ER Roller
Good Survey Tows	number of tows	number of tows	number of tows
Doorspread	1	2	0
Wingspread	1	17	69
Bottom contact	15	5	13
Temperature	78	14	63
Depth	41	21	73
Speed over ground	108	21	73
Total Survey Tows	*109*	*21*	*74*
Depletion Tows			
Doorspread	0	0	0
Wingspread	18	0	0
Bottom contact	21	1	2
Temperature	21	0	0
Depth	21	6	11
Speed over ground	21	6	11
Total Depletion Tows	*21*	*6*	*12*
Mensuration Tows			
Doorspread	7		
Wingspread	9		
Bottom contact	11		
Temperature	12		
Depth	12		
Speed over ground	15		
Total Mensuration Tows	*15*		

Table A13. Summary statistics, 2009 Cooperative Monkfish Survey based on good survey tows and all depletion tows.

		Management Area		
Survey Towns		North	South	North + South
Number of survey tows[1]				
	Endurance cookie	3	18	21
	Endurance roller	64	10	74
	Mary K cookie	0	109	109
	Total	67	137	204
Depth (m)	min-max (median)	30-259 (157)	23-504 (80)	
Number caught		666	2,384	3,050
Kg caught		1,053	5,799	6,852
Number per tow	min-max (median)	0-49 (4)	0-143 (4)	0-143 (4)
Kg per tow	min-max (median)	0-121.5 (5.4)	0-402.2 (7.0)	0-402 (6.8)
Length (cm)	number measured	666	1500	2166
	min-max (median)	13-103 (40)	13-112 (52)	13-112 (49)
Number maturity and gender samples		666	1500	2166

Table A14. Efficiency estimates from 2009 depletion experiments. Gray-shaded estimates were not used in developing population estimates from cooperative survey data.

Cookie Sweep

Exp#	Vessel	Estimate	elower	eupper
1	MK	0.343	0.256	0.472
2	MK	0.950	0.727	1.480
3	MK	0.545	0.368	0.750
4	MK	0.682	0.526	0.846
Average (1, 3, 4)		0.523		

Exp#	Vessel	Estimate	elower	eupper
5	ER	0.382	0.265	0.550
7	ER	0.116	0.079	0.167
Average		0.249		

Roller Sweep

Exp#	Vessel	Estimate	elower	eupper
6	ER	0.050	0.039	0.064
8	ER	0.050	0.038	0.063

Table A15. Comparison of minimum population estimates from 2001, 2004, and 2009 cooperative surveys.

A. Minimum Estimates (assuming 100% net efficiency)

		Nominal Minimum		Sensor Minimum			
		Biomass (mt)	Numbers ('000)	Biomass (mt)	Numbers ('000)	Survey Dates	Percent Zero Tows
2001	North	32,589	25,047	31,454	24,183	Feb. 26- Apr 6	7.9
	South	39,255	22,617	32,622	19,070	Feb. 26- Apr 6	7.6
	N+S	71,844	47,664	64,076	43,253		
2004	North	28,227	14,283	25,583	12,941	~March 1-June 16	10.5
	South	67,879	37,485	61,340	33,971	~March 1-June 16	8.7
	N+S	96,105	51,768	86,923	46,911		
2009	North	12,581	7,951	13,549	8,555	Feb 10 - Apr 17	23.9
	South	28,739	12,693	27,092	11,995	Feb 11 - Apr 26	24.1
	N+S	41,320	20,644	40,642	20,550		

B. Point estimates of population number and biomass assuming intermediate net efficiency.

		Nominal Tow Duration		Sensor Tow Duration	
		Biomass (mt)	Numbers ('000)	Biomass (mt)	Numbers ('000)
2001	North	68,680	52,834	68,680	52,834
	South	66,230	38,037	55,400	32,228
	N+S	134,910	90,870	124,081	85,062
2004	North	86,627	44,053	78,474	39,896
	South	142,410	80,130	128,712	72,614
	N+S	229,037	124,183	207,186	112,510
2009	North	54,916	34,709	59,142	37,345
	South	58,960	25,733	56,398	24,584
	N+S	113,876	60,442	115,540	61,929

Table A16. Absolute biomass estimates from cooperative surveys based on bootstrapping analysis.

Year	Area	Biomass (mt)	Standard Deviation	25th Percentile	Median	75th Percentile
2001	North	80,316	32,512	57,229	74,099	96,238
2001	South	97,475	39,458	69,458	89,921	116,803
2001	N+S	177,791	71,970	126,687	164,020	213,041
2004	North	63,050	23,204	46,591	58,777	74,588
2004	South	182,554	67,187	134,908	170,169	215,922
2004	N+S	245,605	90,391	181,499	228,946	290,510
2009	North	31,451	9,643	24,559	29,921	36,590
2009	South	67,447	20,679	52,663	64,170	78,473
2009	N+S	98,899	30,323	77,222	94,091	115,063

Table A17. Parameters of length-weight equations for monkfish from 2009 cooperative survey and earlier studies. Regression model used was *logW = log a+b log L* where W = weight in kg, L = length in cm.

		Males	Females	Total
2009 Cooperative Survey				
North				
Number of samples		304	356	666
Length range (cm)		13 - 74	13 - 103	13 - 103
Parameter estimates	log(*a*)	-4.613	-4.840	-4.7638
	std err	0.0418	0.0328	0.0259
	b	**2.864**	**3.013**	**2.9627**
	std err	0.0265	0.0202	0.0161
South				
Number of samples		915	567	1498
Length range (cm)		17 - 71	17 - 112	17 - 112
Parameter estimates	log(*a*)	-4.532	-4.799	-4.6846
	std err	0.0234	0.0285	0.0190
	b	**2.834**	**3.011**	**2.9315**
	std err	0.0138	0.0168	0.0112
North + South				
Number of samples		1219	923	2164
Length range (cm)		13 - 74	13 - 112	13 - 112
Parameter estimates	log(*a*)	-4.630	-4.855	-4.7566
	std err	0.0196	0.0219	0.0150
	b	**2.888**	**3.036**	**2.9694**
	std err	0.0118	0.0131	0.0090
DPWG (2007) SCALE model				
North + South (Spring)	log(*a*)			-10.8461
	b			2.9468
Richards et al. 2008				
North + South (Spring)				
Number of samples		2913	3229	
Length range (cm)		40 - 85	40 - 110	
Parameter estimates*****	log(*a*)	-1.4165	-2.0180	
	std err	0.0464	0.0339	
	b	**2.7604**	**3.1228**	
	std err	0.0271	0.0190	

***** weight in grams

Table A18. Maturity parameters estimated from 2009 cooperative monkfish survey and from earlier studies.

			Males	Females
2009 Cooperative Survey				
North	Number of samples		304	356
	Length range (cm)			
		a	-22.982	-19.981
		std err	3.2167	2.5656
		b	0.644	0.511
		std err	0.0895	0.0665
		L_{50}	**35.7**	**39.1**
South	Number of samples		915	567
	Length range (cm)			
		a	-13.518	-17.8882
		std err	1.2552	2.1432
		b	0.366	0.426
		std err	0.0328	0.0506
		L_{50}	**36.9**	**42.0**
North + South	Number of samples		1219	923
	Length range (cm)			
		a	-15.243	-17.221
		std err	1.2285	1.4768
		b	0.421	0.428
		std err	0.0336	0.0371
		L_{50}	**36.2**	**40.3**
DPWG (2007) assessment (2001 cooperative survey data)				
North + South		*a*		-8.7508
		b		0.2045
		L_{50}		**42.8**
Richards et al. (2008) (2001, 2004 coop monkfish surveys)				
North + South				
	Number of samples		2156	2463
	Length range (cm)			
	Parameter	*a*	-11.486	-9.056
		b	0.312	0.221
		L_{50}	**36.8**	**41.0**

Table A19. Nominal minimum area swept biomass and population size estimates from spring 2009, cooperative monkfish survey and NEFSC survey on FSV Henry Bigelow.

	Coop Survey		NEFSC	
	mt	# ('000)	mt	# ('000)
North	12,581	7,951	13,790	7,980
South	28,739	12,693	13,429	6,138
N+S	41,320	20,644	27,218	14,118

Table A20. Stratified mean weight (kg), number, individual fish weight, and length (cm) per tow for goosefish from NEFSC offshore research vessel autumn bottom trawl surveys in the northern management region (strata 20-30, 34-40); confidence limits for both the raw index and the indices smoothed using an integrated moving average (theta = 0.45); minimum and maximum lengths; number of fish caught, number of positive tows, and total number of tows completed each year

	Biomass						Abundance													Number	Number of	
	Raw Index			Smoothed Index			Raw Index			Smoothed Index				Length						of	Nonzero	Number
	Mean	L95%	U95%	Mean	L95%	U95%	Mean	L95%	U95%	Mean	L95%	U95%	Ind wt	Min	5%	50%	Mean	95%	Max	Fish	Tows	of Tows
1963	3.821	2.339	5.304	2.948			0.801	0.512	1.090	0.570			4.661	11	14	59	58.3	103	111	86	39	90
1964	1.892	1.030	2.753	2.476			0.392	0.219	0.564	0.453			4.813	21	21	58	59.4	92	102	32	23	87
1965	2.537	1.407	3.667	2.491			0.347	0.230	0.463	0.397			7.279	28	36	70	71.6	96	110	40	30	88
1966	3.382	2.164	4.600	2.476	1.644	3.730	0.511	0.343	0.678	0.380	0.264	0.549	6.527	37	48	73	73.1	90	96	55	33	86
1967	1.226	0.404	2.049	1.996	1.325	3.007	0.189	0.090	0.288	0.299	0.207	0.431	6.504	48	48	69	70.3	91	92	18	14	86
1968	2.050	0.533	3.568	2.232	1.482	3.363	0.286	0.115	0.457	0.320	0.222	0.461	7.170	11	26	72	71.4	105	106	32	16	86
1969	3.757	1.823	5.690	2.644	1.755	3.983	0.418	0.278	0.559	0.369	0.256	0.532	8.839	13	41	78	78.8	101	110	39	30	88
1970	2.281	0.982	3.580	2.472	1.641	3.724	0.395	0.222	0.569	0.391	0.271	0.564	5.849	22	36	67	67.2	90	98	41	21	92
1971	2.928	1.450	4.405	2.440	1.619	3.676	0.491	0.312	0.671	0.411	0.285	0.593	5.864	15	22	69	67.0	97	101	44	27	94
1972	1.420	0.667	2.174	2.130	1.414	3.209	0.319	0.195	0.442	0.384	0.266	0.554	4.354	21	21	61	56.9	97	99	29	22	94
1973	3.183	1.773	4.594	2.442	1.621	3.679	0.514	0.320	0.709	0.406	0.282	0.586	5.992	16	16	58	65.2	109	112	63	29	92
1974	2.063	1.114	3.011	2.343	1.555	3.529	0.313	0.189	0.436	0.367	0.255	0.530	6.362	13	13	69	64.9	109	111	37	23	97
1975	1.726	1.020	2.432	2.448	1.625	3.688	0.298	0.178	0.418	0.369	0.256	0.533	5.721	11	11	60	62.9	97	102	40	27	106
1976	3.387	1.555	5.219	3.235	2.147	4.874	0.423	0.244	0.601	0.429	0.298	0.619	7.620	29	30	71	72.1	106	121	32	24	87
1977	5.568	3.489	7.646	4.146	2.752	6.246	0.626	0.458	0.794	0.504	0.350	0.727	7.167	21	35	73	71.1	107	119	112	56	126
1978	5.109	3.496	6.722	4.357	2.892	6.564	0.579	0.429	0.729	0.511	0.355	0.738	6.728	10	24	70	67.6	104	116	146	78	201
1979	5.116	3.566	6.665	4.114	2.731	6.198	0.474	0.364	0.584	0.477	0.331	0.689	8.887	15	19	77	73.5	103	115	125	78	211
1980	4.458	2.234	6.682	3.355	2.227	5.055	0.535	0.366	0.703	0.448	0.311	0.646	6.266	6	16	66	63.9	101	111	65	39	97
1981	2.004	0.345	1.529	2.260	1.500	3.405	0.406	0.068	0.216	0.373	0.259	0.538	4.399	9	13	55	57.5	93	101	46	30	93
1982	0.936	0.380	1.492	1.651	1.096	2.487	0.142	0.070	0.213	0.293	0.203	0.423	6.606	29	29	71	68.9	97	100	17	14	95
1983	1.617	0.927	2.308	1.766	1.172	2.661	0.470	0.284	0.656	0.375	0.260	0.541	3.415	13	17	54	53.0	88	96	38	27	82
1984	3.010	1.413	4.607	2.004	1.330	3.020	0.483	0.353	0.613	0.412	0.286	0.595	5.803	11	26	63	62.7	102	106	36	29	88
1985	1.441	0.419	2.463	1.731	1.149	2.608	0.369	0.191	0.548	0.408	0.283	0.588	3.965	12	15	55	53.1	101	102	32	23	88
1986	2.354	1.099	3.608	1.691	1.122	2.547	0.604	0.379	0.829	0.431	0.299	0.621	3.670	19	23	52	53.8	82	100	46	26	90
1987	0.873	0.256	1.491	1.322	0.877	1.991	0.264	0.116	0.411	0.363	0.252	0.524	3.324	15	15	53	52.2	92	96	22	15	87
1988	1.525	0.484	2.565	1.366	0.907	2.058	0.313	0.130	0.496	0.379	0.263	0.546	4.859	11	11	53	57.1	92	93	26	17	89
1989	1.403	0.496	2.310	1.311	0.870	1.974	0.428	0.266	0.590	0.449	0.312	0.648	2.569	9	9	39	40.8	93	96	39	25	87
1990	1.058	0.496	1.620	1.201	0.797	1.810	0.593	0.383	0.804	0.551	0.382	0.795	1.415	9	10	25	32.3	72	89	55	35	89
1991	1.253	0.599	1.908	1.199	0.796	1.806	0.576	0.383	0.769	0.643	0.446	0.927	1.715	9	10	31	38.3	83	95	62	33	88
1992	1.116	0.571	1.661	1.161	0.771	1.750	0.938	0.602	1.274	0.808	0.560	1.165	1.183	9	9	26	33.0	79	86	78	37	86
1993	1.133	0.513	1.754	1.155	0.767	1.741	0.989	0.691	1.287	0.917	0.636	1.323	0.894	6	9	20	27.1	71	94	103	45	86
1994	1.046	0.446	1.645	1.165	0.773	1.755	1.351	0.969	1.732	0.991	0.687	1.429	0.668	9	9	19	24.9	55	98	110	51	87
1995	1.711	0.663	2.759	1.262	0.838	1.902	0.922	0.688	1.155	0.869	0.602	1.253	1.724	10	12	34	39.6	84	91	87	40	93
1996	1.091	0.516	1.665	1.115	0.740	1.680	0.630	0.407	0.853	0.732	0.507	1.055	1.688	8	11	38	40.3	63	95	51	30	88
1997	0.751	0.400	1.102	1.000	0.664	1.507	0.498	0.304	0.693	0.681	0.473	0.983	1.335	8	9	35	35.4	70	86	39	27	90
1998	1.020	0.570	1.470	1.087	0.721	1.637	0.609	0.397	0.820	0.784	0.543	1.130	1.531	10	10	30	35.5	68	77	56	38	104
1999	0.895	0.370	1.420	1.233	0.818	1.857	1.084	0.737	1.431	1.068	0.740	1.540	0.716	8	8	22	25.7	58	81	111	44	106
2000	2.529	1.322	3.736	1.734	1.151	2.613	2.398	1.564	3.232	1.439	0.998	2.076	1.032	9	11	25	30.3	70	88	165	43	87
2001	2.071	1.136	3.005	1.893	1.256	2.852	1.620	1.212	2.027	1.377	0.955	1.986	1.144	8	12	31	34.7	65	93	145	50	90
2002	2.320	1.088	3.553	1.944	1.290	2.930	1.283	0.922	1.645	1.181	0.819	1.704	1.423	9	9	34	35.1	65	93	114	45	86
2003	2.723	1.054	4.393	1.774	1.177	2.674	1.067	0.778	1.357	0.959	0.664	1.384	1.695	8	8	40	37.8	73	88	90	39	88
2004	0.626	0.262	0.989	1.213	0.802	1.835	0.516	0.313	0.720	0.724	0.500	1.048	1.227	8	8	21	29.8	68	89	36	24	85
2005	1.623	0.152	3.094	1.294	0.844	1.986	0.595	0.359	0.830	0.687	0.468	1.006	1.686	8	8	24	34.3	79	88	46	29	87
2006	1.042	0.527	1.557	1.186	0.724	1.943	0.764	0.519	1.010	0.717	0.461	1.114	1.346	6	7	33	33.2	69	86	56	37	94
2007	1.198	0.431	1.965				0.638	0.431	0.844				1.680	9	17	31	37.5	77	81	63	32	90
2008	0.992	0.374	1.609				0.782	0.434	1.129				1.240	9	9	27	31.6	68	85	60	27	90
Bigelow, no calibration coefficient applied:																						
2009	4.275	3.238	5.566				3.091	2.536	3.734				1.369	9	9	32	34.5	69	101	257	61	90
Bigelow, calibration coefficient applied:																						
2009	0.530						0.434															

Table A21. Stratified mean weight (kg), number, individual fish weight, and length (cm) per tow for goosefish from NEFSC offshore research vessel spring bottom trawl surveys in the northern management region (strata 20-30, 34-40); confidence limits for both the raw index and the indices smoothed using an integrated moving average (theta = 0.45); minimum and maximum lengths; number of fish caught, number of positive tows, and total number of tows completed each year.

	Biomass						Abundance													Number	Number of	
	Raw Index			Smoothed Index			Raw Index			Smoothed Index				Length						of	Nonzero	Number
	Mean	L95%	U95%	Mean	L95%	U95%	Mean	L95%	U95%	Mean	L95%	U95%	Ind wt	Min	5%	50%	Mean	95%	Max	Fish	Tows	of Tows
1968	1.008	0.298	1.718	1.223			0.168	0.065	0.272	0.193			5.980	50	51	68	70.4	89	90	13	11	86
1969	1.341	0.160	2.523	1.393			0.180	0.045	0.315	0.213			7.453	33	33	71	71.5	99	100	15	10	87
1970	2.021	0.798	3.245	1.626			0.344	0.216	0.472	0.262			5.867	30	30	62	65.4	98	99	32	22	90
1971	1.039	0.439	1.639	1.641	1.088	2.475	0.158	0.072	0.245	0.268	0.176	0.407	6.488	45	53	69	72.6	99	100	20	15	96
1972	4.678	3.048	6.307	2.252	1.493	3.397	0.643	0.453	0.832	0.390	0.257	0.593	7.105	13	39	74	72.7	100	105	59	38	96
1973	1.908	0.956	2.860	1.891	1.254	2.852	0.435	0.184	0.686	0.407	0.268	0.618	4.313	17	26	68	65.7	99	106	91	36	87
1974	1.477	0.863	2.090	1.578	1.047	2.380	0.438	0.315	0.561	0.405	0.267	0.616	3.391	20	23	58	58.3	97	111	86	41	83
1975	0.936	0.596	1.277	1.377	0.913	2.077	0.339	0.228	0.450	0.384	0.253	0.583	2.760	16	19	53	54.0	87	109	73	36	87
1976	2.826	1.691	3.962	1.558	1.033	2.350	0.673	0.469	0.877	0.394	0.260	0.599	3.759	14	20	60	61.5	95	106	158	52	99
1977	1.028	0.578	1.478	1.182	0.783	1.782	0.259	0.159	0.360	0.283	0.186	0.430	3.594	10	31	66	63.4	93	106	61	37	107
1978	0.626	0.340	0.913	0.984	0.652	1.484	0.141	0.095	0.186	0.216	0.142	0.328	4.014	15	19	73	65.5	89	92	37	30	113
1979	0.904	0.284	1.523	1.110	0.736	1.674	0.144	0.102	0.185	0.219	0.144	0.332	4.652	12	14	67	62.5	100	118	48	40	139
1980	1.622	0.787	2.458	1.438	0.953	2.169	0.379	0.270	0.488	0.294	0.194	0.447	3.748	17	22	43	53.3	98	107	84	38	85
1981	1.744	0.913	2.576	1.718	1.139	2.590	0.376	0.282	0.470	0.333	0.219	0.506	4.444	11	21	52	57.7	95	120	95	42	87
1982	3.015	1.273	4.758	2.031	1.346	3.062	0.346	0.155	0.536	0.348	0.229	0.528	8.594	25	36	61	68.8	105	108	33	22	92
1983	1.587	0.530	2.643	1.840	1.220	2.776	0.418	0.191	0.645	0.365	0.240	0.554	3.663	12	13	49	49.9	96	112	34	22	90
1984	1.696	0.596	2.796	1.843	1.222	2.779	0.328	0.181	0.475	0.349	0.230	0.530	4.732	17	19	62	60.8	93	100	26	19	86
1985	2.113	1.094	3.133	1.951	1.294	2.942	0.346	0.199	0.492	0.347	0.229	0.528	6.122	13	13	68	66.9	104	108	25	21	81
1986	2.165	0.960	3.370	1.957	1.298	2.952	0.340	0.200	0.481	0.347	0.229	0.527	6.244	11	14	63	65.4	109	121	30	22	90
1987	1.728	0.726	2.730	1.835	1.217	2.768	0.245	0.138	0.352	0.352	0.232	0.534	7.052	16	16	66	64.2	99	100	21	16	83
1988	2.111	0.906	3.315	1.792	1.188	2.703	0.610	0.398	0.822	0.454	0.299	0.690	3.343	10	20	49	49.8	89	110	43	26	90
1989	1.636	0.639	2.634	1.567	1.039	2.364	0.625	0.321	0.929	0.481	0.317	0.731	2.590	10	11	40	43.2	80	94	48	24	85
1990	1.005	0.366	1.643	1.332	0.883	2.009	0.282	0.157	0.407	0.428	0.281	0.649	3.587	15	18	47	49.1	106	107	25	17	90
1991	1.827	0.478	3.175	1.368	0.907	2.063	0.593	0.374	0.811	0.502	0.331	0.763	2.723	12	15	35	42.3	78	100	48	28	86
1992	0.910	-0.188	2.008	1.157	0.767	1.744	0.492	0.159	0.825	0.528	0.348	0.802	1.793	16	17	35	40.6	82	101	36	20	83
1993	1.202	0.736	1.668	1.149	0.762	1.733	0.684	0.475	0.893	0.582	0.383	0.885	1.695	10	11	44	41.0	71	90	59	27	87
1994	0.948	0.400	1.496	1.107	0.734	1.669	0.452	0.275	0.629	0.576	0.379	0.875	2.159	10	13	40	41.0	83	89	45	24	88
1995	1.752	0.806	2.698	1.183	0.785	1.785	0.984	0.662	1.305	0.671	0.442	1.020	1.817	15	16	33	39.9	73	97	83	39	88
1996	1.006	0.449	1.563	0.972	0.645	1.466	0.668	0.344	0.992	0.605	0.398	0.919	1.466	15	17	41	43.0	60	70	49	20	82
1997	0.560	0.174	0.946	0.780	0.517	1.176	0.339	0.158	0.520	0.510	0.336	0.775	1.595	9	9	36	39.4	75	89	34	19	89
1998	0.485	0.225	0.745	0.782	0.519	1.180	0.414	0.288	0.540	0.566	0.372	0.859	1.065	11	11	19	31.3	67	78	46	33	115
1999	1.225	0.646	1.804	1.081	0.717	1.631	0.824	0.547	1.102	0.774	0.509	1.175	1.389	9	14	31	35.5	71	97	62	33	87
2000	1.438	0.846	2.030	1.375	0.912	2.074	1.128	0.843	1.413	1.014	0.667	1.540	1.236	15	17	29	34.5	75	87	99	42	89
2001	1.970	0.690	3.251	1.696	1.125	2.558	1.686	1.221	2.151	1.237	0.814	1.879	1.109	9	11	24	31.4	75	86	151	50	89
2002	1.996	1.337	2.655	1.892	1.254	2.854	1.756	1.334	2.178	1.225	0.807	1.862	1.105	12	15	34	36.6	60	73	155	50	91
2003	2.383	0.817	3.949	2.036	1.349	3.073	0.811	0.479	1.144	0.953	0.627	1.449	2.304	10	13	42	44.2	69	95	79	30	86
2004	2.285	0.911	3.659	1.971	1.302	2.984	0.910	0.577	1.243	0.826	0.542	1.260	2.494	9	11	48	46.7	81	85	69	36	88
2005	2.057	0.505	3.609	1.728	1.125	2.654	0.708	0.487	0.929	0.672	0.434	1.039	2.050	11	13	48	45.1	68	75	52	31	87
2006	0.930	0.184	1.675	1.347	0.821	2.209	0.367	0.161	0.573	0.527	0.318	0.871	2.533	15	13	43	44.8	72	105	33	23	95
2007	1.647	-0.614	3.908				0.555	0.247	0.864				1.909	11	10	32	36.8	78	85	43	19	86
2008	1.783	0.1834	3.383				0.681	0.392	0.971				1.910	8	16	35	40.8	73	85	61	24	86
Bigelow, no calibration coefficient applied:																						
2009	4.251	2.7992	5.703				2.33	1.796	2.863													
Bigelow, calibration coefficient applied:																						
2009	0.527						0.327															

Table A22. Stratified mean weight (kg), number, individual fish weight, and length (cm) per tow for goosefish from NEFSC shrimp summer surveys in the northern management region (strata 1, 3, 5-8); confidence limits for indices; minimum and maximum lengths; number of fish caught, number of positive tows, and total number of tows completed. (SURVAN version 8.13)

	Biomass			Abundance				Length						Number	Number of	
	Raw Index			Raw Index										of	Nonzero	Number
	Mean	L95%	U95%	Mean	L95%	U95%	Ind wt	Min	5%	50%	Mean	95%	Max	Fish	Tows	of Tows
1991	1.957	1.165	2.749	2.903	2.268	3.538	0.654	11	15	24	27.5	59	96	125	39	43
1992	2.915	1.399	4.431	2.907	2.27	3.544	0.928	11	13	28	31.5	56	78	135	40	45
1993	3.342	1.388	5.297	3.757	2.699	4.814	0.829	7	9	23	27.6	59	102	170	42	46
1994	1.644	0.837	2.452	3.475	2.430	4.520	0.484	5	10	19	24.1	48	95	166	37	43
1995	1.637	0.729	2.544	2.087	1.216	2.958	0.747	11	19	26	31.2	67	76	83	24	35
1996	3.431	1.331	5.530	2.967	2.105	3.830	1.123	13	14	34	34.4	63	90	107	30	32
1997	2.081	1.040	3.122	1.583	1.073	2.093	1.321	11	16	32	37.7	62	73	72	31	40
1998	2.301	0.714	3.888	2.118	1.500	2.735	1.070	12	16	23	31.3	61	77	84	31	35
1999	6.347	4.766	7.928	7.016	5.305	8.727	0.927	8	9	28	30.9	65	82	301	39	42
2000	4.121	2.090	6.152	5.756	4.101	7.412	0.671	11	15	28	30.2	51	82	215	30	35
2001	8.553	4.443	12.662	11.124	8.463	13.786	0.668	11	13	26	29.5	51	85	442	36	36
2002	12.857	9.180	16.535	11.789	9.379	14.198	1.067	11	17	32	35.3	59	94	493	38	38
2003	8.243	4.470	12.015	5.855	4.174	7.535	1.268	3	13	38	37.4	63	87	236	36	37
2004	4.604	3.464	5.744	3.388	2.662	4.113	1.315	11	11	34	35.7	66	75	142	33	35
2005	7.599	5.133	10.064	5.254	4.185	6.323	1.382	9	14	34	37.4	66	89	271	44	46
2006	7.360	3.812	10.908	4.344	3.089	5.598	1.519	7	11	30	37.2	70	89	143	29	29
2007	5.134	1.844	8.423	4.386	3.264	5.507	0.919	9	11	19	28.2	64	79	218	36	43
2008	3.895	2.120	5.671	2.849	2.078	3.620	1.346	10	14	32	36.1	67	82	116	31	37
2009	4.229	1.519	6.939	3.099	2.361	3.837	1.030	11	13	30	32.7	60	80	159	45	49

Table A23. Monkfish indices from Maine-New Hampshire surveys, strata 1-4.

Year	Fall Stratified Mean Number	SE	Fall Stratified Mean Weight	SE
2000	4.8	0.6	1.65	0.28
2001	11.1	1.6	4.83	0.50
2002	4.1	1.1	3.45	1.14
2003	3.7	0.6	3.60	0.80
2004	3.0	0.5	3.63	0.84
2005	1.8	0.2	2.04	0.47
2006	2.9	0.3	1.79	0.20
2007	3.1	0.4	2.13	0.35
2008	4.1	0.7	2.96	0.41
2009	2.0	0.4	1.93	0.52

Year	Spring Stratified Mean Number	SE	Spring Stratified Mean Weight	SE
2001	6.0	0.91	0.99	0.15
2002	2.4	0.33	1.12	0.17
2003	1.0	0.14	0.64	0.18
2004	1.4	0.17	0.41	0.12
2005	1.1	0.16	0.79	0.15
2006	0.3	0.06	0.15	0.03
2007	1.1	0.18	0.38	0.10
2008	1.37	0.19	0.49	0.08
2009	0.79	0.11	0.20	0.04

Table A24. Stratified mean weight (kg), number, individual fish weight, and length (cm) per tow for goosefish from NEFSC offshore research vessel autumn bottom trawl surveys in the southern management region (strata 1-19, 61-76); confidence limits for both the raw index and the indices smoothed using an integrated moving average (theta = 0.45); minimum and maximum lengths; number of fish caught, number of positive tows, and total number of tows completed each year.

	Biomass						Abundance							Length						Number	Number of	
	Raw Index			Smoothed Index			Raw Index			Smoothed Index										of	Nonzero	Number
	Mean	L95%	U95%	Mean	L95%	U95%	Mean	L95%	U95%	Mean	L95%	U95%	Ind wt	Min	5%	50%	Mean	95%	Max	Fish	Tows	of Tows
1963	3.642	1.818	5.466	4.237			1.197	0.737	1.656	1.270			2.969	7	17	53	50.4	91	97	102	36	73
1964	6.139	2.667	9.612	4.691			1.637	0.907	2.366	1.322			3.482	14	21	53	52.0	86	101	132	34	83
1965	5.093	2.907	7.279	4.335			1.148	0.778	1.519	1.192			4.247	10	15	59	56.3	91	104	83	39	85
1966	7.060	5.062	9.057	3.594	2.156	5.991	1.926	1.364	2.488	1.102	0.650	1.870	3.607	7	7	51	49.6	87	98	101	56	87
1967	1.151	0.623	1.679	1.893	1.136	3.155	0.519	0.324	0.715	0.700	0.413	1.188	2.195	14	19	31	40.6	83	100	98	42	163
1968	0.904	0.461	1.346	1.393	0.836	2.322	0.399	0.206	0.591	0.544	0.321	0.923	2.211	12	17	45	46.3	75	86	77	39	164
1969	1.360	0.506	2.214	1.370	0.822	2.284	0.537	0.308	0.766	0.520	0.307	0.883	2.466	10	14	41	45.4	88	96	101	43	163
1970	1.340	0.643	2.037	1.355	0.813	2.258	0.350	0.235	0.466	0.487	0.287	0.827	3.632	4	13	55	53.3	84	104	58	35	161
1971	0.711	0.282	1.139	1.350	0.810	2.250	0.282	0.150	0.414	0.570	0.336	0.967	2.788	5	8	39	42.3	95	98	55	28	168
1972	5.045	3.374	6.716	2.068	1.241	3.447	4.113	1.281	6.944	1.070	0.631	1.816	1.298	12	16	23	31.8	74	99	604	85	161
1973	2.030	1.036	3.025	1.740	1.044	2.901	1.176	0.857	1.495	0.813	0.479	1.379	1.568	13	14	32	37.7	77	93	280	70	154
1974	0.710	0.322	1.098	1.320	0.792	2.201	0.218	0.116	0.320	0.482	0.284	0.817	3.277	14	16	54	52.9	81	101	56	26	153
1975	2.050	1.333	2.767	1.519	0.912	2.533	0.653	0.434	0.871	0.487	0.287	0.825	2.653	8	17	45	46.3	87	105	127	51	158
1976	1.093	0.547	1.639	1.430	0.858	2.384	0.314	0.189	0.438	0.403	0.238	0.684	3.166	11	11	51	50.7	77	95	60	34	165
1977	1.883	1.203	2.563	1.612	0.967	2.688	0.372	0.265	0.479	0.395	0.233	0.670	4.170	5	16	55	53.1	95	106	94	50	172
1978	1.395	0.883	1.906	1.638	0.982	2.730	0.259	0.178	0.340	0.403	0.238	0.683	4.469	13	17	61	56.5	87	101	68	39	219
1979	2.275	1.278	3.272	1.853	1.112	3.089	0.694	0.483	0.905	0.553	0.326	0.938	2.307	7	16	34	40.5	84	109	182	70	205
1980	1.883	1.181	2.585	1.826	1.096	3.044	0.726	0.427	1.024	0.652	0.384	1.105	2.211	3	16	34	41.6	85	104	113	42	159
1981	2.864	0.889	4.840	1.763	1.058	2.939	0.965	0.579	1.351	0.714	0.421	1.211	1.961	6	17	38	40.7	71	99	176	59	146
1982	0.657	0.361	0.953	1.229	0.737	2.048	0.610	0.373	0.847	0.638	0.376	1.083	1.060	13	15	26	32.5	66	73	98	42	143
1983	2.156	0.700	3.611	1.304	0.782	2.174	0.776	0.470	1.082	0.589	0.347	0.999	2.304	7	16	45	44.4	72	100	109	49	146
1984	0.750	0.158	1.343	0.987	0.592	1.645	0.311	0.114	0.508	0.451	0.266	0.765	2.445	5	13	47	45.7	68	93	42	25	146
1985	1.327	0.761	1.893	0.899	0.539	1.498	0.524	0.356	0.692	0.443	0.261	0.752	2.055	17	17	40	42.0	72	96	100	46	145
1986	0.561	0.245	0.877	0.630	0.378	1.049	0.325	0.169	0.481	0.389	0.229	0.660	1.523	7	14	34	37.6	68	78	60	33	146
1987	0.276	0.118	0.433	0.477	0.286	0.794	0.482	0.308	0.657	0.385	0.227	0.654	0.575	12	13	20	25.0	56	61	67	27	132
1988	0.554	0.210	0.898	0.521	0.312	0.868	0.230	0.097	0.364	0.328	0.194	0.557	2.376	19	27	36	45.1	87	91	27	19	129
1989	0.642	0.300	0.985	0.546	0.328	0.910	0.382	0.182	0.582	0.356	0.210	0.603	1.366	7	7	42	38.0	57	77	57	23	129
1990	0.445	0.047	0.844	0.514	0.308	0.856	0.294	0.115	0.472	0.367	0.216	0.623	1.050	9	13	24	33.1	61	81	47	22	136
1991	0.797	0.244	1.349	0.532	0.319	0.886	0.690	0.248	1.133	0.440	0.259	0.746	0.901	14	15	23	30.8	57	81	106	27	131
1992	0.318	0.193	0.444	0.419	0.252	0.699	0.342	0.223	0.461	0.390	0.230	0.661	0.919	8	11	30	32.2	54	74	46	21	129
1993	0.295	0.058	0.532	0.399	0.239	0.664	0.290	0.136	0.444	0.377	0.222	0.639	0.784	10	13	32	30.4	52	68	46	24	130
1994	0.620	0.190	1.050	0.464	0.278	0.773	0.598	0.353	0.843	0.434	0.256	0.737	0.906	8	12	25	29.2	59	83	85	31	135
1995	0.413	0.186	0.640	0.443	0.266	0.739	0.493	0.259	0.727	0.404	0.238	0.685	0.777	11	13	25	29.4	54	66	72	29	129
1996	0.387	0.217	0.557	0.445	0.267	0.741	0.235	0.132	0.338	0.329	0.194	0.557	1.638	18	19	42	42.3	62	68	31	21	131
1997	0.592	0.354	0.829	0.490	0.294	0.816	0.308	0.198	0.418	0.335	0.197	0.568	1.914	9	9	49	44.6	70	71	43	24	131
1998	0.500	0.244	0.756	0.475	0.285	0.792	0.332	0.150	0.514	0.361	0.213	0.612	1.525	11	11	36	37.0	68	87	45	20	131
1999	0.304	0.196	0.412	0.445	0.267	0.741	0.450	0.319	0.582	0.410	0.242	0.696	0.672	12	14	27	29.2	52	55	109	44	106
2000	0.485	0.269	0.700	0.538	0.323	0.896	0.422	0.270	0.575	0.439	0.259	0.745	1.102	5	15	33	34.3	63	70	64	30	132
2001	0.712	0.373	1.050	0.696	0.418	1.161	0.378	0.239	0.518	0.483	0.285	0.819	1.724	4	11	39	41.69	70	80	51	30	130
2002	1.315	0.785	1.846	0.889	0.533	1.482	0.829	0.565	1.092	0.626	0.369	1.062	1.514	6	14	41	39.12	61	81	110	47	130
2003	0.827	0.542	1.112	0.872	0.523	1.455	0.951	0.627	1.276	0.671	0.395	1.139	0.858	6	7	18	28.25	59	70	128	41	130
2004	0.969	0.332	1.606	0.886	0.529	1.485	0.474	0.247	0.702	0.569	0.334	0.970	1.598	7	15	45	40.36	64	78	67	32	133
2005	0.804	0.409	1.198	0.849	0.498	1.447	0.575	0.339	0.811	0.546	0.314	0.949	1.309	7	13	42	38.47	57	67	76	34	123
2006	0.834	0.379	1.288	0.843	0.456	1.559	0.452	0.280	0.624	0.506	0.268	0.956	1.660	6	12	44	40.6	65	77	83	36	151
2007	0.505	0.247	0.764				0.195	0.106	0.284				2.571	25	25	51	50.1	68	69	27	19	142
2008	0.412	0.112	0.712				0.198	0.098	0.305				2.076	4	4	45	38.6	69	88	39	20	142
Bigelow, no calibration coefficient applied:																						
2009	1.524	1.303	1.767				1.417	1.197	1.658				1.2	6	7	63	33.4	27	77	351	85	176
Bigelow, calibration coefficient applied:																						
2009	0.189						0.199															

Table A25. Stratified mean weight (kg), number, individual fish weight, and length (cm) per tow for goosefish from NEFSC offshore research vessel spring bottom trawl surveys in the southern management region (strata 1-19, 61-76); confidence limits for both the raw index and the indices smoothed using an integrated moving average (theta = 0.45); minimum and maximum lengths; number of fish caught, number of positive tows, and total number of tows completed each year. Data prior to 1971 has been revised following an audit of historical data and the data reflect an increase in precision in the calculations of delta distributions. (SAGA version 3.55)

	Biomass						Abundance													Number	Number of	
	Raw Index			Smoothed Index			Raw Index			Smoothed Index				Length						of	Nonzero	Number
	Mean	L95%	U95%	Mean	L95%	U95%	Mean	L95%	U95%	Mean	L95%	U95%	Ind wt	Min	5%	50%	Mean	95%	Max	Fish	Tows	of Tows
1968	1.159	0.568	1.750	1.083			0.212	0.126	0.297	0.217			5.414	21	23	63	62.5	94	95	65	31	150
1969	0.955	0.444	1.466	1.034			0.221	0.138	0.305	0.220			4.097	7	25	47	54.3	91	111	41	31	155
1970	1.009	0.465	1.553	1.042			0.176	0.104	0.248	0.223			5.648	22	22	65	63.9	102	108	40	31	166
1971	0.769	0.322	1.216	1.072	0.653	1.761	0.204	0.105	0.304	0.264	0.173	0.403	3.675	13	16	50	53.3	101	115	42	24	160
1972	1.892	1.172	2.612	1.379	0.840	2.265	0.364	0.266	0.461	0.373	0.244	0.569	5.169	14	22	59	59.1	103	123	79	48	165
1973	1.897	1.539	2.255	1.435	0.874	2.357	1.051	0.854	1.249	0.534	0.350	0.816	2.172	11	19	32	41.1	80	110	589	128	187
1974	1.164	0.769	1.560	1.238	0.754	2.032	0.486	0.369	0.604	0.486	0.318	0.742	3.236	14	21	44	49.1	93	117	201	70	132
1975	0.947	0.574	1.320	1.112	0.677	1.827	0.447	0.326	0.568	0.441	0.289	0.674	2.795	10	22	44	47.6	87	107	169	61	134
1976	1.209	0.833	1.585	1.114	0.678	1.829	0.404	0.307	0.500	0.397	0.260	0.607	3.340	13	22	48	51.5	91	110	259	78	162
1977	1.205	0.771	1.640	1.055	0.642	1.733	0.299	0.231	0.367	0.354	0.232	0.540	4.607	16	21	51	56.8	95	116	173	75	160
1978	0.745	0.522	0.968	0.914	0.557	1.501	0.335	0.265	0.405	0.353	0.231	0.538	2.986	11	17	39	45.9	90	104	196	66	161
1979	0.757	0.464	1.051	0.908	0.553	1.492	0.281	0.164	0.397	0.364	0.238	0.555	2.944	10	14	37	44.4	98	124	125	50	194
1980	0.799	0.494	1.104	1.021	0.621	1.676	0.451	0.355	0.548	0.446	0.292	0.681	1.926	18	21	34	40.8	83	106	346	99	204
1981	1.816	1.157	2.475	1.351	0.823	2.219	0.784	0.542	1.027	0.543	0.356	0.830	2.563	12	22	40	44.6	89	113	345	74	141
1982	2.810	1.591	4.028	1.467	0.893	2.410	0.942	0.657	1.226	0.517	0.339	0.790	2.324	11	14	38	42.4	89	104	251	68	150
1983	0.955	0.421	1.489	1.029	0.627	1.690	0.270	0.176	0.365	0.329	0.216	0.503	3.514	24	24	47	51.8	97	112	55	36	147
1984	0.748	0.223	1.272	0.759	0.462	1.247	0.182	0.090	0.275	0.239	0.157	0.365	4.067	21	21	47	50.9	96	97	35	22	149
1985	0.327	0.089	0.565	0.565	0.344	0.928	0.159	0.072	0.247	0.209	0.137	0.319	2.052	22	22	39	42.3	85	90	31	21	147
1986	0.832	0.352	1.312	0.608	0.371	0.999	0.283	0.125	0.442	0.219	0.144	0.335	2.917	15	24	43	48.7	90	102	65	36	149
1987	0.496	-0.014	1.007	0.531	0.323	0.871	0.108	0.054	0.162	0.194	0.127	0.296	4.612	15	15	59	52.7	102	103	30	21	150
1988	0.427	0.302	0.552	0.484	0.295	0.795	0.440	0.286	0.595	0.253	0.166	0.387	0.971	17	18	30	34.0	61	82	67	33	132
1989	0.365	0.237	0.493	0.480	0.292	0.789	0.202	0.102	0.302	0.229	0.150	0.349	1.500	15	24	41	41.4	69	79	36	18	129
1990	1.005	0.565	1.445	0.573	0.349	0.941	0.205	0.152	0.258	0.224	0.147	0.343	4.034	16	21	53	56.5	86	93	39	23	128
1991	0.590	0.316	0.865	0.469	0.285	0.770	0.319	0.144	0.494	0.234	0.153	0.357	1.509	15	23	33	37.6	69	101	61	31	132
1992	0.210	0.070	0.350	0.329	0.200	0.540	0.177	0.089	0.266	0.198	0.130	0.302	1.235	14	19	28	35.0	69	85	28	17	128
1993	0.264	0.098	0.430	0.311	0.189	0.511	0.195	0.099	0.292	0.180	0.118	0.275	1.319	17	19	38	38.6	56	72	29	18	128
1994	0.321	0.138	0.504	0.329	0.200	0.540	0.114	0.058	0.170	0.156	0.102	0.238	2.379	13	13	41	44	91	93	24	18	131
1995	0.526	0.032	1.020	0.353	0.215	0.579	0.196	0.109	0.283	0.166	0.109	0.254	2.637	18	19	38	46	80	81	32	20	129
1996	0.286	0.146	0.426	0.289	0.176	0.475	0.135	0.075	0.196	0.158	0.104	0.242	2.083	9	9	44	44	80	81	27	20	143
1997	0.132	0.071	0.193	0.239	0.146	0.393	0.124	0.070	0.177	0.168	0.110	0.256	1.064	18	18	37	36	58	75	38	14	130
1998	0.282	0.190	0.374	0.295	0.180	0.485	0.254	0.175	0.333	0.218	0.143	0.333	1.110	12	16	35	36	64	77	40	30	131
1999	0.629	0.375	0.883	0.376	0.229	0.618	0.335	0.229	0.441	0.256	0.168	0.391	1.899	16	19	41	43	74	94	63	32	131
2000	0.294	0.179	0.408	0.339	0.206	0.556	0.242	0.155	0.329	0.250	0.164	0.382	1.222	14	14	38	38	61	78	32	25	131
2001	0.243	0.094	0.393	0.336	0.204	0.551	0.234	0.136	0.332	0.251	0.164	0.383	1.092	11	15	34	36	57	68	44	50	89
2002	0.375	0.134	0.616	0.413	0.252	0.679	0.318	0.096	0.540	0.263	0.172	0.401	1.181	22	23	37	39	53	62	50	50	91
2003	1.423	0.894	1.953	0.543	0.330	0.892	0.308	0.200	0.415	0.242	0.158	0.369	3.721	15	29	57	57	80	87	65	30	86
2004	0.193	0.061	0.324	0.373	0.226	0.616	0.116	0.055	0.178	0.189	0.123	0.290	1.565	22	21	37	40	61	62	24	36	88
2005	0.369	0.234	0.504	0.399	0.238	0.671	0.259	0.111	0.407	0.206	0.132	0.320	1.424	20	20	36	39	61	68	41	26	131
2006	0.540	0.216	0.863	0.451	0.248	0.819	0.172	0.097	0.247	0.191	0.115	0.319	3.136	24	15	37	53	80	80	28	20	132
2007	0.559	0.295	0.823				0.259	0.172	0.345				2.136	20	23	48	46	69	75	77	30	158
2008	0.3866	0.137	0.636				0.1887	0.0731	0.3044				2.064	17	17	41	46	64	84	32	19	140
Bigelow, no calibration coefficient applied:																						
2009	3.0167	1.467	4.566				1.1726	0.8171	1.5281													
Bigelow, calibration coefficient applied:																						
2009	0.374						0.164															

Table A26. Stratified mean weight (kg), number, individual fish weight, and length (cm) per tow for goosefish from NEFSC winter flatfish surveys in the southern management region (strata 1-3, 5-7, 9-11, 13-14, 61-63, 65-67, 69-71, 73-75); confidence limits for indices; minimum and maximum lengths; number of fish caught, number of positive tows, and total number of tows completed. The last survey in this time series was completed in 2007.

	Biomass Raw Index			Abundance Raw Index				Length						Number of	Number of	
	Mean	L95%	U95%	Mean	L95%	U95%	Ind wt	Min	5%	50%	Mean	95%	Max	Fish	Nonzero Tows	Number of Tows
1992	6.314	4.160	8.468	5.234	3.854	6.614	1.139	11	22	33	36.0	51	95	582	66	100
1993	6.357	4.563	8.150	4.952	3.898	6.005	1.193	9	21	36	37.7	53	98	555	77	108
1994	3.321	2.372	4.270	2.484	1.870	3.097	1.298	8	16	31	35.1	61	78	278	56	77
1995	3.774	2.472	5.076	3.137	2.104	4.170	1.209	19	21	35	37.4	57	101	365	76	106
1996	4.496	3.435	5.557	3.438	2.662	4.213	1.294	10	22	37	39.1	57	100	456	87	119
1997	4.460	3.190	5.731	2.976	2.323	3.629	1.456	10	18	39	39.8	59	82	359	89	107
1998	2.849	1.997	3.701	1.494	1.150	1.838	1.876	10	20	41	44.1	69	103	203	77	114
1999	4.090	3.066	5.114	3.068	2.370	3.767	1.319	10	17	34	37.8	61	87	362	83	115
2000	5.690	4.023	7.356	4.428	3.166	5.689	1.265	11	24	103	39.2	103	96	616	93	118
2001	7.182	4.501	9.863	4.380	2.997	5.762	1.383	8	24	103	39.3	103	84	729	115	142
2002	6.235	4.794	7.675	3.474	2.737	4.212	1.744	15	30	103	44.5	103	86	550	113	143
2003	5.482	3.491	7.473	2.258	1.580	2.937	2.418	12	25	103	45.5	103	85	316	72	86
2004	7.171	4.308	10.034	4.397	2.836	5.957	1.568	13	23	103	41.2	103	88	682	103	123
2005	4.531	2.657	6.405	2.972	2.043	3.902	1.497	13	23	103	40.0	103	90	313	59	91
2006	5.481	4.022	6.939	3.082	2.327	3.837	1.743	22	31	103	44.7	103	92	430	78	114
2007	3.395	2.586	4.205	1.472	1.212	1.732	2.251	14	23	42	48.3	103	91	217	83	118

Table A27. Stratified mean number and length (cm) per tow for goosefish from NEFSC summer scallop surveys in the southern management region (shellfish strata 6, 7, 10, 11, 14, 15, 18, 19, 22-31, 33-35, 46, 47, 55, 58-61, 621, 631); confidence limits for the raw index using an integrated moving average (theta = 0.45); minimum and maximum lengths; number of fish caught, number of positive tows, and the total number of tows completed each year. (SURVAN version 8.13)

	Abundance														
	Raw Index			Smoothed Index			Length						Number of Fish	Number of Nonzero Tows	Number of Tows
	Mean	L95%	U95%	Mean	L95%	U95%	Min	5%	50%	Mean	95%	Max			
1984	1.285	1.109	1.461				6	11	28	29.5	54	82	410	165	254
1985	1.521	1.256	1.786				7	9	25	28.7	53	84	493	183	282
1986	1.246	1.045	1.446				8	10	15	22.9	54	95	431	183	296
1987	3.152	2.767	3.537				8	9	13	18.6	51	90	1253	255	315
1988	1.666	1.385	1.947				7	12	28	29.8	49	97	572	187	316
1989	0.995	0.833	1.156				6	10	31	31.9	53	101	303	147	304
1990	1.534	1.339	1.729				6	10	18	24.4	54	94	563	205	303
1991	2.284	1.994	2.574				7	9	14	21.0	45	94	808	241	315
1992	1.939	1.661	2.217				5	9	25	27.3	52	97	644	235	316
1993	2.845	2.568	3.123				8	10	15	21.8	48	73	995	258	301
1994	3.401	3.006	3.796				8	10	15	22.2	51	87	1145	265	314
1995	2.263	1.968	2.558				7	9	27	29.6	57	92	764	243	314
1996	2.005	1.746	2.265				7	9	23	29.9	59	81	638	226	298
1997	1.110	0.954	1.265				7	13	33	36.7	65	76	388	196	313
1998	1.014	0.876	1.152				6	11	20	30.2	61	79	371	183	319
1999	2.592	2.161	3.022				6	10	16	23.5	55	84	856	248	306
2000	2.242	1.973	2.510				8	9	18	27.3	54	87	832	240	315
2001	1.710	1.484	1.936				7	8	35	36.0	64	77	549	233	334
2002	1.711	1.488	1.933				7	11	35	34.2	60	86	598	203	310
2003	2.784	2.394	3.174				6	9	15	24.4	58	87	819	211	294
2004	2.875	2.506	3.244				9	11	26	29.8	61	83	860	290	348
2005	2.013	1.753	2.274				8	10	28	31.3	56	83	859	265	344
2006	1.445	1.272	1.618				7	7	29	31.1	61	83	571	230	327
2007	0.8272	0.6938	0.9606				7	12	39	40.2	69	84	366	183	336
2008	1.0024	0.8283	1.1765				7	7	26	31.297	68	75	350	162	285
2009	0.7858	0.6341	0.9375				6	10	25	30.9	65	80	248	133	269

Table A28. Stratified mean weight (kg), number, individual fish weight, and length (cm) per tow for goosefish from NEFSC offshore research vessel autumn bottom trawl surveys in the northern and southern management regions; confidence limits for both the raw index and the indices smoothed using an integrated moving average (theta = 0.45); minimum and maximum lengths; number of fish caught, number of positive tows, and total number of tows completed each year.

	Biomass			Abundance										Number	Number of	
	Raw Index			Raw Index				Length						of	Nonzero	Number
	Mean	L95%	U95%	Mean	L95%	U95%	Ind wt	Min	5%	50%	Mean	95%	Max	Fish	Tows	of Tows
1963	7.4	3.046	11.75	0.993	0.725	1.261	7.951	7	16	55	53.9	96	111	188	75	164
1964	3.822	2.846	4.798	0.985	0.626	1.343	3.994	14	20	54	53.5	89	102	164	57	170
1965	4.627	2.924	6.331	0.728	0.542	0.915	6.433	10	19	62	60.1	93	110	123	69	173
1966	5.3	4.137	6.464	1.185	0.903	1.466	4.42	7	8	57	55.0	89	98	214	88	169
1967	2.027	1.148	2.907	0.381	0.26	0.501	5.578	14	19	41	46.8	91	100	116	56	250
1968	2.697	1.224	4.169	0.351	0.219	0.484	7.913	11	20	53	54.8	89	106	109	55	250
1969	3.291	1.884	4.697	0.487	0.342	0.632	7.024	10	16	56	56.9	95	110	134	70	240
1970	3.341	1.731	4.952	0.369	0.27	0.468	8.895	4	17	58	59.5	90	104	99	56	251
1971	3.529	1.309	5.749	0.37	0.262	0.477	8.715	5	9	58	56.1	95	101	99	55	262
1972	8.911	5.512	12.31	2.52	0.876	4.164	4.464	12	16	23	33.1	75	99	633	107	252
1973	4.34	2.018	6.662	0.898	0.696	1.1	4.769	13	15	36	44.3	92	112	343	99	246
1974	2.014	0.945	3.084	0.258	0.179	0.337	7.69	13	14	63	59.0	97	111	93	49	250
1975	2.763	1.736	3.791	0.504	0.368	0.64	5.385	8	17	50	50.4	89	105	167	78	264
1976	2.103	1.265	2.941	0.359	0.255	0.464	5.504	11	27	62	61.3	94	121	92	58	252
1977	3.445	2.487	4.403	0.479	0.385	0.573	7.05	5	19	64	63.0	99	119	206	106	298
1978	2.987	2.247	3.727	0.393	0.315	0.472	7.159	10	18	65	63.4	99	116	214	117	420
1979	3.562	2.659	4.465	0.604	0.471	0.736	5.338	7	16	47	51.1	97	115	307	148	416
1980	3.115	2.056	4.174	0.645	0.458	0.832	4.667	3	16	40	49.4	98	111	178	81	256
1981	2.705	1.469	3.94	0.73	0.501	0.96	3.244	6	17	42	44.6	80	101	222	89	239
1982	0.885	0.516	1.254	0.414	0.273	0.554	2.142	13	15	32	37.7	75	100	115	56	238
1983	2.214	1.18	3.248	0.651	0.455	0.847	3.123	7	16	48	47.0	79	100	147	76	228
1984	1.9	1.112	2.689	0.383	0.257	0.51	4.825	5	13	56	54.7	93	106	78	54	234
1985	1.548	0.915	2.18	0.459	0.336	0.582	3.456	12	17	44	45.7	88	102	132	69	233
1986	1.827	0.708	2.947	0.442	0.311	0.573	4.018	7	17	43	46.9	81	100	106	59	236
1987	0.541	0.267	0.816	0.392	0.273	0.511	1.383	12	14	22	32.6	65	96	89	42	219
1988	0.957	0.48	1.433	0.265	0.156	0.374	3.607	11	23	46	51.0	89	93	53	36	218
1989	1.419	0.707	2.132	0.401	0.266	0.536	3.49	7	8	41	39.2	84	96	96	48	216
1990	1.295	0.71	1.879	0.418	0.282	0.554	3.034	9	10	25	32.6	70	89	102	57	225
1991	1.536	0.837	2.235	0.643	0.372	0.914	2.294	9	13	27	33.6	69	95	168	60	219
1992	1.08	0.562	1.597	0.59	0.434	0.746	1.886	8	8	27	32.7	72	86	124	58	215
1993	1.777	0.813	2.74	0.58	0.427	0.733	2.752	6	9	22	28.1	56	94	149	69	216
1994	1.512	0.636	2.389	0.91	0.697	1.124	1.523	8	10	21	26.5	56	98	195	82	222
1995	1.429	0.655	2.203	0.671	0.503	0.838	2.039	10	13	33	35.2	69	91	159	69	222
1996	0.781	0.445	1.117	0.399	0.288	0.509	1.946	8	14	40	41.0	63	95	82	51	219
1997	1.135	0.662	1.607	0.387	0.284	0.49	2.913	8	9	40	39.7	70	86	82	51	221
1998	1	0.634	1.367	0.447	0.309	0.585	2.199	10	10	30	36.2	68	87	101	58	235
1999	1.051	0.498	1.603	0.713	0.55	0.876	1.265	8	9	23	27.1	54	81	220	80	236
2000	1.656	1.027	2.285	1.242	0.885	1.599	1.315	5	11	25	31.1	65	88	229	77	219
2001	1.276	0.84	1.711	0.894	0.706	1.081	1.289	4	11	32	36.4	65	93	196	80	220
2002	1.732	1.134	2.33	1.017	0.802	1.232	1.466	6	10	37	37.0	63	93	224	92	216
2003	1.614	0.902	2.327	0.999	0.775	1.224	1.227	6	8	25	32.5	62	88	218	80	218
2004	0.827	0.424	1.229	0.492	0.334	0.649	1.434	7	8	29	35.7	66	89	103	56	218
2005	1.144	0.491	1.798	0.583	0.414	0.752	1.468	7	8	32	36.7	66	88	122	63	217
2006	0.92	0.579	1.261	0.582	0.438	0.725	1.49	6	7	38	36.6	65	86	139	74	245
2007	0.793	0.441	1.145	0.379	0.279	0.479	1.949	9	17	36	41.3	77	90	89	51	232
2008	0.652	0.342	0.963	0.44	0.284	0.596	1.458	4	5	29	33.5	68	88	100	47	232
2009	2.949	2.129	3.769	2.166	1.79	2.541	1.288	6	9	30	34.1	68	101	608	146	266

Table A29. Stratified mean weight (kg), number, individual fish weight, and length (cm) per tow for goosefish from NEFSC offshore research vessel spring bottom trawl surveys in the northern and southern management regions; confidence limits for both the raw index and the indices smoothed using an integrated moving average (theta = 0.45); minimum and maximum lengths; number of fish caught, number of positive tows, and total number of tows completed each year.

	Biomass			Abundance										Number	Number of	
	Raw Index			Raw Index				Length						of	Nonzero	Number
	Mean	L95%	U95%	Mean	L95%	U95%	Ind wt	Min	5%	50%	Mean	95%	Max	Fish	Tows	of Tows
1968	1.501	0.586	2.417	0.193	0.127	0.259	7.704	21	23	63	65.2	89	95	78	42	238
1969	1.139	0.56	1.718	0.204	0.129	0.278	5.458	7	21	63	61.2	95	111	56	41	242
1970	1.774	0.871	2.676	0.247	0.178	0.315	7.167	22	25	62	64.7	98	108	72	53	255
1971	0.948	0.573	1.322	0.185	0.117	0.253	5.061	13	20	58	60.2	99	115	62	39	257
1972	3.857	2.679	5.035	0.481	0.383	0.578	7.898	13	25	67	66.8	100	123	138	84	259
1973	2.629	1.862	3.397	0.792	0.637	0.948	3.667	11	20	41	46.8	88	110	680	164	274
1974	2.198	1.281	3.114	0.466	0.381	0.551	5.162	14	22	46	52.7	93	117	287	111	215
1975	1.301	0.85	1.751	0.402	0.318	0.487	3.449	10	21	47	49.8	87	109	242	97	221
1976	1.888	1.364	2.412	0.517	0.414	0.619	3.31	13	21	56	57.0	93	110	417	130	261
1977	1.152	0.835	1.469	0.284	0.226	0.342	3.796	10	23	58	59.5	93	116	234	113	268
1978	0.71	0.529	0.891	0.253	0.209	0.298	2.674	11	17	45	50.4	89	104	233	96	273
1979	0.951	0.587	1.315	0.221	0.152	0.291	3.66	10	14	42	49.3	99	123	173	90	333
1980	1.144	0.752	1.537	0.421	0.348	0.494	2.439	17	21	37	45.6	89	107	430	137	289
1981	1.786	1.268	2.303	0.612	0.466	0.759	2.832	11	22	42	48.0	93	120	440	116	228
1982	3.002	1.962	4.042	0.691	0.508	0.875	4.189	11	17	44	47.9	99	108	284	90	242
1983	1.22	0.679	1.761	0.332	0.222	0.442	3.593	12	19	49	50.8	96	112	89	58	237
1984	1.146	0.593	1.699	0.243	0.162	0.325	4.445	17	20	58	56.5	93	100	61	41	235
1985	1.754	0.956	2.552	0.238	0.158	0.317	7.387	13	21	55	57.3	104	108	56	42	228
1986	1.592	0.96	2.224	0.307	0.198	0.417	5.202	11	20	54	56.5	99	121	95	58	239
1987	1.115	0.561	1.669	0.165	0.11	0.219	6.774	15	15	65	59.8	99	103	51	37	233
1988	1.126	0.621	1.632	0.511	0.384	0.637	2.146	10	19	34	41.8	80	110	110	59	222
1989	1.181	0.531	1.831	0.377	0.238	0.516	2.945	10	11	40	42.6	74	94	84	42	214
1990	1.224	0.657	1.792	0.237	0.177	0.297	5.156	15	18	49	52.8	92	107	64	40	217
1991	1.48	0.665	2.295	0.432	0.295	0.569	3.087	12	15	33	40.2	78	101	109	59	218
1992	0.754	0.149	1.36	0.307	0.16	0.453	2.461	14	17	33	38.7	82	101	64	37	211
1993	1.082	0.584	1.58	0.399	0.295	0.502	2.838	10	12	42	40.3	71	90	88	45	215
1994	0.844	0.401	1.288	0.255	0.174	0.335	3.315	10	13	40	41.8	83	93	69	42	219
1995	1.371	0.679	2.064	0.523	0.38	0.665	2.744	15	16	34	41.2	75	97	115	59	217
1996	0.647	0.388	0.906	0.356	0.217	0.495	1.783	9	15	43	43.2	67	81	76	40	225
1997	0.408	0.225	0.591	0.214	0.132	0.295	1.925	9	11	36	38.2	75	89	72	33	219
1998	0.677	0.194	1.159	0.32	0.251	0.39	2.089	11	12	30	33.4	66	78	86	63	246
1999	1.085	0.585	1.584	0.535	0.406	0.665	2.068	9	15	32	38.2	71	97	125	65	218
2000	0.85	0.558	1.143	0.609	0.481	0.738	1.373	14	16	31	35.3	70	87	131	67	220
2001	0.96	0.422	1.497	0.836	0.635	1.037	1.106	9	12	27	32.0	71	86	195	76	220
2002	1.047	0.74	1.355	0.914	0.696	1.132	1.121	12	16	35	37.1	58	73	205	73	222
2003	1.821	1.102	2.541	0.517	0.365	0.668	3.064	10	14	47	48.5	74	95	144	57	211
2004	1.06	0.485	1.635	0.445	0.303	0.588	2.351	9	6	99	45.6	117	85	93	48	219
2005	1.069	0.421	1.717	0.445	0.319	0.571	1.839	11	8	100	43.1	115	75	93	57	218
2006	0.702	0.339	1.064	0.253	0.157	0.349	2.773	16	5	101	48.3	115	105	61	43	227
2007	1.01	0.06	1.96	0.382	0.244	0.519	2	11	7	99	40.6	117	85	120	49	244
2008	0.966	0.287	1.645	0.393	0.255	0.531	1.954	8	4	101	42.4	116	85	93	43	226
2009	3.529	2.441	4.618	1.653	1.349	1.957	1.885	11	5	101	42.0	115	93	0	127	297

Table A30. Age length key used for estimating mean lengths at age and variation from ages in the spring, winter, 2001 & 2004 cooperative, and fall surveys.

	age										
length	1	2	3	4	5	6	7	8	9	10	total
8	1										1
9	4										4
10	19										19
11	25	3									28
12	26	9									35
13	23	21									44
14	24	18									42
15	27	28									55
16	15	48									63
17	22	43									65
18	26	56	2								84
19	8	54	16								78
20	4	50	34								88
21		25	72								97
22		29	82								111
23		32	81	1							114
24		22	120								142
25		23	127								150
26		27	149								176
27		22	174	5							201
28		20	140	53							213
29		6	89	130							225
30		4	46	163							213
31		3	26	178							207
32			26	183							209
33			22	154							176
34		1	19	192							212
35			23	203							226
36			25	184							209
37			20	197	6						223
38			20	173	31						224
39			11	104	84						199
40			8	63	140						211
41			3	29	171						203
42				26	200						226
43			1	22	209						232
44				26	197						223
45				19	200						219
46				24	179						203
47				28	184	4					216
48				17	197	32					246
49				12	123	81					216
50				13	98	141					252
51				2	33	157					192
52				1	28	186					215
53					24	186					210
54					20	184					204
55					19	198					217
56					15	191	1				207
57					12	179	1				192
58					20	143	3				166
59					19	117	25				161
60					8	68	87				163
61					2	37	99				138
62						19	113				132
63					1	13	81				95
64						9	101				110
65						12	86				98
66						7	60				67
67						5	63				68
68						3	66				69
69						8	53	2			63
70						3	38	23			64
71						3	27	32			62
72							16	52			68
73							2	52			54
74							4	51			55
75							1	38			39
76							4	42			46
77							4	31			35
78							2	41			43
79							1	26			27
80							3	40	9		52
81							2	18	9		29
82							1	18	20		39
83								5	20		25
84								2	25		27
85								2	18		20
86								3	10	1	14
87								1	15		16
88								4	12		16
89								2	7		9
90									2	1	3
91									7		7
92									3	2	5
93									4		4
94									2		2
95								1	2	2	5
96									1	2	3
97									2		2
98									1	1	2
102										2	2
103										1	1
105										2	2
107										1	1
110										1	1
total	224	544	1336	2202	2220	1986	944	486	169	16	10127

Table A31. Area swept expansions used for scaling the stratified numbers per tow indices. Nm^2 represents the square nautical miles covered by the survey.

Survey	nm^2	footprint	expansions
Shrimp North	6,147	0.00350	1,756,286
Winter South	30,014	0.01270	2,363,307
Scallop South	13,204	0.00110	12,003,636
Fall & Spring North	26,265	0.01120	2,345,089
Fall & Spring South	37,081	0.01120	3,310,804
Fall and spring combine albatross	63,346	0.01120	5,655,893
Fall and spring combine Bigelow	63,346	0.00700	9,049,429
ME/NH Fall North	4,517	0.00462	977,324
MDMF Fall North	1,055	0.00385	274,311

Table A32. Northern area goosefish SCALE runs residual sum of squares, input weights & effective sample sizes, estimated Qs, Fstart, age-1 recruitment in year 1 (1980), and estimated logistic selectivity parameters (L_{50}, slope). First column under each run=weights, second column=residual sum of square.

Run number	2007		1		2		3		4		5		6		7		8	
Discription	Data Poor		add 07-09		add ME/NH		drop MDMF		est Fstart		high eff samp		low vrec		high vrec		Final WG run	
	Final run				and MDMF						catch lf						fix Fstart	
total objective function		241.34		263.77		428.14		289.85		288.54		509.84		267.04		338.43		291.22
total catch	10	0.68	10	1.67	10	4.12	10	3.27	10	3.53	10	11.49	10	4.69	10	1.10	10	3.57
catch len freq 1+	400	9.57	400	11.67	400	13.50	400	12.24	400	12.39	10k	207.35	400	14.43	400	11.31	400	12.35
Vrec	5	24.93	5	24.72	5	29.14	5	28.09	5	28.01	5	24.37	2	18.86	25	29.00	5	28.02
Fall age 1	2	32.41	2	35.19	2	35.45	2	35.15	2	34.83	2	43.23	2	29.25	2	50.20	2	34.69
Spring age 2	2	29.45	2	31.78	2	29.71	2	30.28	2	30.19	2	34.39	2	24.32	2	45.29	2	29.35
Spring age 3	2	30.78	2	31.79	2	31.68	2	31.55	2	31.79	2	34.76	2	29.69	2	40.59	2	32.16
Shrimp age 1	2	21.54	2	25.74	2	28.04	2	25.81	2	25.72	2	30.63	2	25.70	2	25.63	2	26.49
Shrimp age 2	2	6.52	2	7.13	2	6.15	2	6.46	2	6.46	2	11.14	2	6.03	2	9.23	2	6.35
Fall ME/NH age 1					2	15.92	2	16.40	2	16.38	2	20.67	2	15.33	2	20.14	2	15.76
Fall MDMF age 1					2	16.11												
Fall adult 40+	3	15.96	3	15.68	3	13.74	3	14.80	3	14.52	3	12.98	3	14.54	3	14.41	3	15.17
Spring adult 40+	3	12.84	3	14.00	3	12.84	3	13.09	3	11.33	3	9.59	3	11.41	3	12.65	3	14.32
Shrimp adult 40+	3	15.11	3	17.83	3	20.59	3	18.25	3	18.48	3	15.09	3	17.69	3	22.02	3	18.60
Fall ME/NH 40+					3	3.00	3	3.51	3	3.33	3	3.90	3	2.73	3	5.51	3	3.35
Fall MDMF 40+					3	24.30												
Fall len freq 30+	25	13.82	25	14.78	25	15.16	25	14.94	25	15.26	25	15.00	25	15.44	25	14.75	25	14.96
Spring len freq 30+	25	13.18	25	14.21	25	14.60	25	14.37	25	14.67	25	14.59	25	14.90	25	14.12	25	14.40
Shrimp len freq 30+	75	14.28	75	15.85	75	16.29	75	15.91	75	15.95	75	15.11	75	16.25	75	16.24	75	15.95
Coop len freq 30+	100	0.26	100	0.49	100	0.68	100	0.57	100	0.60	100	0.45	100	0.70	100	0.56	100	0.58
Fall Bigelow len freq 30+				0.72	100	0.90	100	0.78	100	0.80	100	0.90	100	0.86	100	0.68	100	0.79
Spring Bigelow len freq 30+				0.51	100	0.62	100	0.54	100	0.55	100	0.62	100	0.59	100	0.50	100	0.55
Fall ME/NH len freq 30+					50	3.60	50	3.85	50	3.74	50	3.61	50	3.61	50	4.51	50	3.81
Fall MDMF len freq 30+					50	91.99												
Q Fall age 1		0.024		0.009		0.011		0.010		0.010		0.011		0.010		0.007		0.010
Q Spring age 2		0.036		0.008		0.010		0.009		0.009		0.010		0.010		0.007		0.009
Q Spring age 3		0.049		0.014		0.017		0.016		0.016		0.017		0.017		0.012		0.016
Q Shrimp age 1		0.025		0.034		0.041		0.038		0.039		0.040		0.038		0.031		0.040
Q Shrimp age 2		0.038		0.098		0.116		0.109		0.110		0.115		0.108		0.088		0.112
Q ME/NH age 1						0.015		0.013		0.013		0.015		0.013		0.011		0.014
Q MDMF age 1						0.001												
Q Fall adult 40+		0.041		0.040		0.054		0.047		0.052		0.055		0.058		0.029		0.048
Q Spring adult 40+		0.044		0.043		0.059		0.051		0.056		0.060		0.062		0.032		0.052
Q Shrimp adult 40+		0.130		0.107		0.156		0.129		0.138		0.147		0.151		0.079		0.134
Q ME/NH adult 40+						0.066		0.051		0.055		0.057		0.059		0.033		0.054
Q MDMF adult 40+						0.003												
Fstart		0.01		0.01		0.01		0.01		0.36		0.39		0.36		0.12		0.01
recruitment year 1		20.5		18.8		15.7		16.5		16.6		15.9		14.3		25.7		16.1
Selectivity																		
block 1 (1980-2009)																		
alpha		42.7		43.1		56.5		47.7		49.3		49.2		56.4		39.2		48.9
beta		0.16		0.16		0.12		0.14		0.13		0.14		0.12		0.19		0.13

Table A33. Southern area goosefish SCALE runs residual sum of squares, input weights & effective sample sizes, estimated Qs, Fstart, age1 recruitment in year 1, and the estimated logistic selectivity parameters (L_{50}, slope). First column under each run are weights, residual sum of squares in the second .

Run number	2007		1		2		3		4		5		6		7		8	
Discription	Data Poor Final run		add 07-09		est Fstart		1 block		2 block		3 block		high eff samp catch lf		higher eff samp catch lf		Final run 2 block, fix Fstart	
total objective function		287.71		357.68		348.72		353.23		348.78		348.32		390.70		450.93		358.77
total catch	10	0.93	10	0.96	10	0.50	10	0.59	10	0.50	10	0.48	10	1.22	10	4.69	10	0.91
catch len freq 1+	400	9.22	400	12.12	400	12.33	400	13.22	400	12.53	400	11.67	2k	45.68	5k	91.47	400	12.09
Vrec	5	13.59	5	20.97	5	20.35	5	20.63	5	20.35	5	20.40	5	18.49	5	17.39	5	22.00
Fall age 1	2	29.50	2	49.64	2	49.10	2	49.34	2	49.14	2	48.99	2	51.08	2	53.65	2	49.34
Spring age 2	2	16.95	2	33.49	2	33.45	2	33.36	2	33.45	2	33.46	2	33.69	2	34.02	2	33.79
Spring age 3	2	36.32	2	40.27	2	40.15	2	40.41	2	40.16	2	40.14	2	41.03	2	42.46	2	40.00
Winter age 2	2	6.85	2	6.65	2	6.62	2	6.70	2	6.61	2	6.63	2	7.06	2	7.83	2	6.67
Winter age 3	2	12.27	2	13.21	2	12.98	2	13.24	2	12.97	2	12.98	2	12.69	2	12.92	2	13.03
Scallop age 1	3	29.31	3	33.14	3	32.87	3	33.08	3	32.89	3	32.82	3	35.12	3	37.69	3	32.55
Scallop age 2	3	13.56	3	16.39	3	16.06	3	16.27	3	16.07	3	16.03	3	17.11	3	18.70	3	15.95
Fall adult 40+	3	20.74	3	22.84	3	20.14	3	20.93	3	20.03	3	20.28	3	20.73	3	21.45	3	24.44
Spring adult 40+	3	27.87	3	28.86	3	24.54	3	26.43	3	24.46	3	24.75	3	24.65	3	25.36	3	28.82
winter adult 40+	3	4.08	3	5.18	3	5.01	3	5.18	3	5.00	3	5.08	3	5.21	3	5.37	3	5.25
Scallop adult 40+	3	16.66	3	17.42	3	17.73	3	16.78	3	17.76	3	17.65	3	19.28	3	20.93	3	17.36
fall len freq 30+	25	12.60	25	13.92	25	14.04	25	13.99	25	14.04	25	14.04	25	13.90	25	13.86	25	13.91
spring len freq 30+	25	16.84	25	17.98	25	18.02	25	18.00	25	18.02	25	18.01	25	18.02	25	18.10	25	17.97
winter len freq 30+	75	5.64	75	6.43	75	6.53	75	6.56	75	6.52	75	6.55	75	6.25	75	6.17	75	6.43
Coop len freq 30+	100	0.33	100	0.71	100	0.73	100	0.75	100	0.72	100	0.73	200	1.36	200	1.33	100	0.72
Scallop len freq 30+	75	14.46	75	16.37	75	16.42	75	16.61	75	16.41	75	16.47	75	16.01	75	15.67	75	16.40
Fall Bigelow len freq 30+			100	0.70	100	0.71	100	0.72	100	0.71	100	0.71	200	1.34	200	1.19	100	0.70
Spring Bigelow len freq 30+			100	0.42	100	0.44	100	0.44	100	0.43	100	0.44	200	0.80	200	0.68	100	0.43
Q Fall age 1		0.024		0.006		0.005		0.005		0.005		0.005		0.006		0.008		0.006
Q Spring age 2		0.045		0.002		0.002		0.002		0.002		0.002		0.002		0.003		0.002
Q Spring age 3		0.045		0.010		0.008		0.008		0.008		0.008		0.010		0.012		0.009
Q Winter age 2		0.038		0.010		0.008		0.008		0.008		0.008		0.010		0.012		0.010
Q Winter age 3		0.046		0.086		0.072		0.072		0.073		0.069		0.084		0.100		0.083
Q Scallop age 1		0.026		0.286		0.237		0.240		0.241		0.229		0.280		0.334		0.281
Q Scallop age 2		0.040		0.172		0.142		0.144		0.145		0.138		0.168		0.201		0.168
Q Fall adult 40+		0.027		0.025		0.020		0.019		0.021		0.019		0.025		0.033		0.023
Q Spring adult 40+		0.018		0.017		0.014		0.013		0.014		0.013		0.017		0.023		0.016
Q winter adult 40+		0.249		0.162		0.137		0.127		0.140		0.130		0.174		0.229		0.155
Q Scallop adult 40+		0.510		0.196		0.164		0.155		0.168		0.157		0.205		0.267		0.187
Fstart		0.2		0.2		0		0		0		0		0		0		0.2
recruitment year 1		31.05		27.4		32.2		32.3		31.7		33.4		26.9		31.5		28.1
Selectivity																		
block 1		80-95		80-95		80-95		80-09		80-95		80-95		80-95		80-95		80-01
alpha		40.238		47.802		25.83		43.99		25.87		25.81		32.33		37.13		45.59
beta		0.1304		0.1017		0.28		0.15		0.28		0.28		0.19		0.15		0.15
block 2		96-03		96-03		96-03				96-09		96-01		96-01		96-01		02-09
alpha		48.323		48.495		46.21				47.68		42.63		42.45		44.12		50.69
beta		0.1469		0.1456		0.16				0.15		0.19		0.18		0.17		0.13
block 3		04-07		04-07		04-07						02-07		02-07		02-07		
alpha		50.981		50.13		48.77						49.70		52.78		60		
beta		0.134		0.14		0.14						0.14		0.13		0.1196		

Table A34. Combined management area goosefish runs of residual sum of squares, input weights & effective sample sizes, estimated Qs, Fstart, age1 recruitment in year 1, and the estimated logistic selectivity parameters (L_{50}, slope). First column under each run are the weights. Residual sum of squares are in the second column.

Run number	1		2		3		4	
Discription	1 block combined		2 block		high eff samp catch lf		Final run add ME/NH	
total objective function		324.83		324.70		386.14		356.83
total catch	10	0.89	10	0.86	10	4.63	10	1.73
catch len freq 1+	400	8.60	400	8.36	4k	57.50	400	8.42
Vrec	5	16.05	5	15.90	5	15.06	5	17.59
Fall age 1	2	23.93	2	23.96	2	26.42	2	24.48
Spring age 2	2	26.71	2	26.82	2	27.86	2	25.05
Spring age 3	2	23.15	2	23.19	2	25.74	2	23.70
Winter age 2	2	8.97	2	8.96	2	9.39	2	9.41
Winter age 3	2	15.00	2	14.97	2	15.10	2	14.98
Scallop age 1	2	28.03	2	27.97	2	29.53	2	28.58
Scallop age 2	2	16.52	2	16.46	2	17.81	2	16.76
Shrimp age 1	2	31.58	2	31.51	2	31.84	2	32.18
Shrimp age 2	2	9.34	2	9.35	2	10.01	2	8.66
Fall ME/NH age 1							2	18.42
Fall adult 40+	3	10.55	3	10.66	3	9.56	3	11.81
Spring adult 40+	3	11.01	3	11.08	3	10.25	3	11.70
winter adult 40+	3	5.48	3	5.53	3	4.68	3	5.32
Scallop adult 40+	3	12.47	3	12.49	3	14.17	3	12.56
Shrimp adult 40+	3	21.06	3	21.11	3	22.03	3	21.12
Fall ME/NH 40+							3	4.77
fall len freq 30+	25	7.27	25	7.27	25	7.20	25	7.29
spring len freq 30+	25	8.28	25	8.28	25	8.38	25	8.33
winter len freq 30+	75	6.50	75	6.52	75	6.13	75	6.46
Coop len freq 30+	100	0.44	100	0.45	100	0.80	100	0.44
Scallop len freq 30+	75	15.63	75	15.64	75	15.08	75	15.44
Fall Bigelow len freq 30+	100	0.38	100	0.38	100	0.70	100	0.39
Spring Bigelow len freq 30	100	0.28	100	0.28	100	0.51	100	0.29
Shrimp len freq 30+	75	16.69	75	16.70	75	15.76	75	16.43
Fall ME/NH len freq 30+							50	4.51
Q Fall age 1		0.007		0.007		0.010		0.008
Q Spring age 2		0.004		0.004		0.005		0.005
Q Spring age 3		0.011		0.011		0.015		0.013
Q Winter age 2		0.004		0.004		0.005		0.005
Q Winter age 3		0.036		0.035		0.049		0.043
Q Scallop age 1		0.120		0.116		0.161		0.145
Q Scallop age 2		0.073		0.071		0.098		0.088
Q Shrimp age 1		0.014		0.014		0.019		0.017
Q Shrimp age 2		0.038		0.037		0.051		0.046
Q ME/NH age 1								0.006
Q Fall adult 40+		0.024		0.023		0.037		0.030
Q Spring adult 40+		0.022		0.021		0.034		0.027
Q winter adult 40+		0.068		0.066		0.111		0.086
Q Scallop adult 40+		0.088		0.085		0.139		0.111
Q Shrimp adult 40+		0.032		0.031		0.052		0.041
Q ME/NH adult 40+								0.017
Fstart		0		0		0		0.1
recruitment year 1		56.2		58.2		40.2		47.0
Selectivity								
block 1		80-09		80-01		80-01		80-01
alpha		41.02		39.16		42.53		43.73
beta		0.17		0.20		0.16		0.16
block 2				02-09		02-09		02-09
alpha				42.27		49.78		42.34
beta				0.16		0.14		0.16

Table A35. Estimates of age-1 recruitment, biomass and fishing mortality rates from SCALE model final runs. Estimates by area do not sum to combined area because combined data were fit independently to the SCALE model.

North					South					North+South				
Year	Age-1 Recruitment (millions)	Exploitable Biomass (kt)	Total Biomass (kt)	F	Year	Age-1 Recruitment (millions)	Exploitable Biomass (kt)	Total Biomass (kt)	F	Year	Age-1 Recruitment (millions)	Exploitable Biomass (kt)	Total Biomass (kt)	F
1980	16.10	82.46	100.41	0.06	1980	28.15	81.96	107.06	0.09	1980	47.01	185.69	224.35	0.07
1981	11.46	78.75	96.30	0.06	1981	29.97	89.37	115.01	0.06	1981	40.21	190.56	229.05	0.06
1982	11.91	75.72	92.72	0.07	1982	24.06	98.43	124.27	0.05	1982	33.52	197.14	234.75	0.06
1983	11.63	72.64	88.83	0.08	1983	21.67	107.29	132.90	0.05	1983	29.94	202.77	238.44	0.06
1984	10.63	70.08	85.11	0.09	1984	21.24	114.83	139.68	0.05	1984	31.26	206.82	239.60	0.06
1985	8.18	67.05	80.64	0.11	1985	20.38	121.29	144.76	0.05	1985	30.39	207.98	237.52	0.07
1986	11.94	62.73	75.29	0.10	1986	23.54	124.26	146.10	0.04	1986	36.68	202.91	230.28	0.06
1987	11.17	58.72	70.65	0.13	1987	35.80	125.30	146.74	0.04	1987	53.11	195.62	223.31	0.07
1988	13.62	53.13	64.71	0.16	1988	9.86	123.74	144.90	0.05	1988	26.26	184.10	212.73	0.08
1989	18.60	47.13	59.08	0.23	1989	25.47	119.42	141.06	0.12	1989	47.96	170.54	201.39	0.15
1990	21.67	39.86	53.05	0.26	1990	33.10	108.17	130.58	0.10	1990	60.76	150.32	184.00	0.14
1991	17.09	34.17	48.97	0.28	1991	38.97	101.92	124.95	0.14	1991	60.04	138.62	174.57	0.17
1992	18.94	30.59	47.29	0.35	1992	31.91	93.72	117.58	0.21	1992	55.46	128.64	167.51	0.23
1993	29.38	29.19	47.97	0.59	1993	43.44	82.27	109.37	0.28	1993	82.42	120.00	164.34	0.34
1994	26.59	26.01	45.86	0.60	1994	35.18	73.93	104.02	0.25	1994	69.22	111.59	159.72	0.30
1995	12.33	24.98	45.20	0.75	1995	29.46	73.01	104.47	0.31	1995	44.06	113.24	163.06	0.35
1996	15.79	22.70	43.10	0.89	1996	22.94	72.48	103.37	0.32	1996	36.24	114.10	163.68	0.37
1997	28.49	20.93	41.24	0.71	1997	24.03	73.69	102.86	0.33	1997	54.61	117.86	164.65	0.32
1998	34.25	22.80	42.80	0.42	1998	42.71	74.33	101.37	0.32	1998	87.62	125.04	168.26	0.25
1999	44.00	27.41	49.04	0.42	1999	37.69	73.38	99.18	0.26	1999	90.31	132.81	176.67	0.21
2000	44.14	30.09	56.03	0.46	2000	33.29	75.61	102.21	0.17	2000	89.04	140.18	190.91	0.18
2001	29.07	32.00	63.18	0.68	2001	16.24	80.07	108.54	0.21	2001	51.68	149.19	208.16	0.24
2002	18.41	31.86	65.53	0.82	2002	32.18	75.42	111.90	0.20	2002	50.72	159.36	217.60	0.22
2003	18.77	32.88	65.46	1.13	2003	41.83	79.97	117.06	0.22	2003	59.05	172.86	227.53	0.25
2004	19.80	30.01	57.08	0.96	2004	24.29	84.23	119.19	0.16	2004	48.59	181.57	228.84	0.19
2005	14.75	28.98	50.61	0.71	2005	16.46	89.88	123.05	0.16	2005	31.23	189.35	230.49	0.17
2006	25.03	29.00	47.89	0.38	2006	14.45	92.91	125.72	0.13	2006	49.41	195.11	233.33	0.12
2007	18.37	32.74	51.41	0.22	2007	13.11	97.80	129.20	0.12	2007	33.19	207.81	243.26	0.09
2008	17.46	38.96	58.23	0.14	2008	17.88	103.98	131.09	0.10	2008	38.97	219.86	252.43	0.07
2009	16.15	46.15	66.06	0.10	2009	18.99	108.74	131.22	0.07	2009	35.74	224.32	255.33	0.05

Table A36. Calculated age-specific retrospective adjustments based on 7 peels.

	AGE											
area	1	2	3	4	5	6	7	8	9	10	11	12
North	78%	74%	73%	71%	71%	64%	51%	39%	29%	23%	19%	17%
South	71%	89%	92%	92%	92%	90%	88%	86%	85%	83%	81%	78%
Combined	76%	79%	80%	79%	80%	78%	75%	73%	71%	69%	66%	63%

Table A37. Summary of possible explanations for lack of fit and/or retrospective error in SCALE model results.

Error type	**Observation**	**Hypothesis for Observation**	**Perceived Likelihood**	**Evidence For or Against**
Observation Error	Recruitment pulse in North late 1990s	Caused by change in survey q	Low	NO: -Multiple surveys show pulse -shows up in CPUE at plausible lags -No reason to expect Q change YES: -Discarding did not show major increase
	Declining / not-increasing survey indices	Caused by change in survey q	Low	NO: - multiple surveys show trend - no changes in survey gear or method until 2009
	Declining / not-increasing survey indices	Caused by change in availability of monks to survey	Low	NO: -survey timing has not changed in recent years (except scallop 2009) -habitat compression due to climate change not seen in GoM or Northern MAB
	Catch has declined	Due to more than change in regulations	Low	NO: -reporting methods haven't changed recent years -recent discard sampling rates decent
	Catch and survey LF's do not expand when catches decline	Fish move out of survey / fishery area	???	NO: -Scotian Shelf summer indices have same trend as US North MAYBE: -monkfish do occur in deeper water (at least ~900 m) but not necessarily just large ones

	Catch and survey LF’s do not expand when catches decline	Larger fish do not grow rapidly (aging method wrong)	Possibly high	See below (Growth model wrong)
Process Error	Growth is linear	Growth model wrong	Possibly high	-Age method has not been validated -Other Lophius: some show curvature in growth curve -European studies: early growth faster than previously thought
		M wrong	High	Probably live longer than we give them credit for (max obs size = 138 cm, max size aged = 113 cm = 13 yr) If age method missing annuli, then they live longer
		Emigration	Med-High	YES: Patterns in sex ratio at length suggest portion of the stock (maturing females) absent from the US shelf at least some parts of the year NO: Scotian Shelf survey indices show same trends as US North

Table A38. Results of age-based yield-per-recruit analysis using M=0.3 and area-specific selectivity patterns estimated by SCALE model. A-B: 2006 analysis, C-E: 2009 analysis.

DPWG

A. North

Reference Point	F	YPR	SSBR	Total B / R
Fzero	0.00	0.00	7.97	9.94
F-01	0.18	0.56	3.22	4.81
F-Max	0.31	0.60	2.06	3.51
F at 40% MSP	0.18	0.56	3.19	4.77

B. South

Reference Point	F	YPR	SSBR	Total B / R
Fzero	0.00	0.00	5.32	6.41
F-01	0.25	0.50	2.43	3.39
F-Max	0.40	0.53	1.72	2.61
F at 40% MSP	0.31	0.52	2.13	3.06

SAW50

C. North

Reference Point	F	YPR	SSBR	Total B / R
Fzero	0.00	0.00	5.39	6.41
F-01	0.27	0.51	2.55	3.46
F-Max	0.43	0.54	1.85	2.69
F at 40% MSP	0.35	0.54	2.15	3.03

D. South

Reference Point	F	YPR	SSBR	Total B / R
Fzero	0.00	0.00	5.39	6.41
F-01	0.28	0.52	2.59	3.51
F-Max	0.46	0.55	1.88	2.73
F at 40% MSP	0.38	0.55	2.15	3.04

E. North+South

Reference Point	F	YPR	SSBR	Total B / R
Fzero	0.00	0.00	5.39	6.41
F-01	0.24	0.48	2.44	3.32
F-Max	0.37	0.51	1.74	2.55
F at 40% MSP	0.28	0.50	2.15	3.00

Table A39. Estimated biological reference points, biomass and F for monkfish in northern and southern management regions and areas combined.

Management	Biomass BRPs in metric tons				Estimates			
Areas								
North	**BRP**	**Basis**	**DPSWG 2007**	**SDWG 2010**		**DPSWG 2007**	**SDWG 2010**	**SDWG 2010 Adjust**
	Fmax	YPR	0.31	0.43	Current F	0.09	0.10	0.17
					Current B	119,000	66,062	31,761
	Bthreshold	Bloss 1980-2006	65,200					
	Bthreshold	Bloss 1980-2009		41,238				
	Bthreshold	0.5*Bmax Projected		26,465				
	Bthreshold	0.5*Bmax Proj Adjust		20,643				
	Btarget	Bavg 1980-2006	92,200	62,371				
	Btarget	Bavg 1980-2009		61,991				
	Btarget	Bmax Projected		52,930				
	Btarget	Bmax Proj Adjust		41,286				
	MSY	Fmax Projected		10,745				
South	**BRP**	**Basis**	**DPSWG 2007**	**SDWG 2010**		**DPSWG 2007**	**SDWG 2010**	**SDWG 2010 Adjust**
	Fmax	YPR	0.40	0.46	Current F	0.12	0.07	0.08
					Current B	135,000	131,218	113,119
	Bthreshold	Bloss 1980-2006	96,400					
	Bthreshold	Bloss 1980-2009		99,181				
	Bthreshold	0.5*Bmax Projected		37,245				
	Bthreshold	0.5*Bmax Proj Adjust		28,461				
	Btarget	Bavg 1980-2006	122,500	120,292				
	Btarget	Bavg 1980-2009		121,313				
	Btarget	Bmax Projected		74,490				
	Btarget	Bmax Proj Adjust		56,922				
	MSY	Fmax Projected		15,279				
Combined	**BRP**	**Basis**	**DPSWG 2007**	**SDWG 2010**		**DPSWG 2007**	**SDWG 2010**	**SDWG 2010 Adjust**
	Fmax	YPR		0.37	Current F		0.05	0.06
					Current B		255,326	186,369
	Bthreshold	Bloss 1980-2009		159,715				
	Bthreshold	0.5*Bmax Projected		64,501				
	Bthreshold	0.5*Bmax Proj Adjust		49,021				
	Btarget	Bavg 1980-2009		208,190				
	Btarget	Bmax Projected		129,002				
	Btarget	Bmax Proj Adjust		98,041				
	MSY	Fmax Projected		25,943				

Table A40. Projected catch and biomass (mt) for northern management region.

NMA Projection Table: Catch and Biomass in Metric tons
Annual P relative to BRP **n/a = not applicable**

ACT

Year	F	Total Catch	Total Biomass	P < 0.5*Bmax	P < Bloss2006	P < Bloss2009	P > Fmax
2010	0.10	4,447	74,102	0%	5%	0%	0%
2011	0.22	10,750	81,907	0%	0%	0%	0%
2012	0.22	10,750	81,204	0%	1%	0%	0%
2013	0.22	10,750	80,225	0%	2%	0%	0%
2014	0.23	10,750	78,944	0%	4%	0%	0%
2015	0.24	10,750	77,548	0%	8%	0%	0%
2016	0.24	10,750	76,383	0%	14%	0%	0%

ABC

Year	F	Total Catch	Total Biomass	P < 0.5*Bmax	P < Bloss2006	P < Bloss2009	P > Fmax
2010	0.10	4,447	74,102	0%	3%	0%	0%
2011	0.38	17,485	81,907	0%	0%	0%	4%
2012	0.44	17,485	73,769	0%	4%	0%	52%
2013	0.54	17,485	64,796	0%	52%	0%	94%
2014	0.71	17,485	55,815	0%	86%	1%	99%
2015	1.01	17,485	46,871	0%	96%	26%	100%
2016	1.69	17,485	37,631	12%	99%	72%	100%

Fthreshold

Year	F	Total Catch	Total Biomass	P < 0.5*Bmax	P < Bloss2006	P < Bloss2009	P > Fmax
2010	0.10	4,447	74,102	0%	5%	0%	0%
2011	0.43	19,557	81,907	0%	0%	0%	n/a
2012	0.43	16,553	70,831	0%	12%	1%	n/a
2013	0.43	14,120	62,846	0%	68%	44%	n/a
2014	0.43	12,402	57,627	0%	89%	73%	n/a
2015	0.43	11,384	54,619	0%	93%	80%	n/a
2016	0.43	10,883	53,298	0%	93%	84%	n/a

Table A41. Projected catch and biomass (mt) for southern management region.

SMA Projection Table: Catch and Biomass in Metric tons							
Annual P relative to BRP			**n/a = not applicable**				
ACT							
Year	**F**	**Total Catch**	**Total Biomass**	**P < 0.5*Bmax**	**P < Bloss2006**	**P < Bloss2009**	**P > Fmax**
2010	0.07	6,235	131,344	0%	0%	0%	0%
2011	0.13	11,469	132,243	0%	0%	0%	0%
2012	0.14	11,469	126,295	0%	0%	0%	0%
2013	0.15	11,469	121,055	0%	1%	1%	0%
2014	0.16	11,469	116,674	0%	2%	4%	0%
2015	0.17	11,469	113,979	0%	5%	8%	0%
2016	0.17	11,469	113,777	0%	7%	11%	0%
ABC							
Year	**F**	**Total Catch**	**Total Biomass**	**P < 0.5*Bmax**	**P < Bloss2006**	**P < Bloss2009**	**P > Fmax**
2010	0.07	6,235	131,344	0%	0%	0%	0%
2011	0.15	13,326	132,243	0%	0%	0%	0%
2012	0.16	13,326	124,255	0%	0%	0%	0%
2013	0.18	13,326	114,149	0%	1%	2%	0%
2014	0.20	13,326	111,160	0%	7%	12%	0%
2015	0.22	13,326	107,047	0%	16%	23%	0%
2016	0.23	13,326	105,443	0%	22%	30%	0%
Fthreshold							
Year	**F**	**Total Catch**	**Total Biomass**	**P < 0.5*Bmax**	**P < Bloss2006**	**P < Bloss2009**	**P > Fmax**
2010	0.07	6,235	131,344	0%	0%	0%	0%
2011	0.46	36,245	132,243	0%	0%	0%	n/a
2012	0.46	25,171	99,182	0%	33%	50%	n/a
2013	0.46	18,484	80,735	0%	99%	100%	n/a
2014	0.46	15,033	72,167	0%	100%	100%	n/a
2015	0.46	13,857	69,597	0%	100%	100%	n/a
2016	0.46	13,878	69,949	0%	100%	100%	n/a

Table A42. Projected catch and biomass (mt) for northern and southern management regions combined.

Combined Management Areas Projection Table: Catch and Biomass in Metric tons							
Annual P relative to BRP			**n/a = not applicable**				
ACT							
Year	**F**	**Total Catch**	**Total Biomass**	**P < 0.5*Bmax**	**P < Bloss2006**	**P < Bloss2009**	**P > Fmax**
2010	0.05	9,903	254,702	0%	n/a	0%	0%
2011	0.12	22,219	259,839	0%	n/a	0%	0%
2012	0.13	22,219	248,386	0%	n/a	0%	0%
2013	0.14	22,219	238,189	0%	n/a	0%	0%
2014	0.15	22,219	229,182	0%	n/a	0%	0%
2015	0.16	22,219	222,237	0%	n/a	0%	0%
2016	0.16	22,219	218,434	0%	n/a	0%	0%
ABC							
Year	**F**	**Total Catch**	**Total Biomass**	**P < 0.5*Bmax**	**P < Bloss2006**	**P < Bloss2009**	**P > Fmax**
2010	0.05	9,903	254,702	0%	n/a	0%	0%
2011	0.17	30,811	259,839	0%	n/a	0%	0%
2012	0.19	30,811	238,818	0%	n/a	0%	0%
2013	0.21	30,811	219,525	0%	n/a	0%	0%
2014	0.24	30,811	202,164	0%	n/a	0%	0%
2015	0.26	30,811	187,460	0%	n/a	7%	0%
2016	0.29	30,811	176,021	0%	n/a	23%	7%
Fthreshold							
Year	**F**	**Total Catch**	**Total Biomass**	**P < 0.5*Bmax**	**P < Bloss2006**	**P < Bloss2009**	**P > Fmax**
2010	0.05	9,903	254,702	0%	n/a	0%	0%
2011	0.37	62,664	259,839	0%	n/a	0%	n/a
2012	0.37	47,163	203,542	0%	n/a	0%	n/a
2013	0.37	36,947	167,133	0%	n/a	25%	n/a
2014	0.37	30,678	145,682	0%	n/a	87%	n/a
2015	0.37	27,411	134,286	0%	n/a	97%	n/a
2016	0.37	26,005	129,290	0%	n/a	98%	n/a

Figures

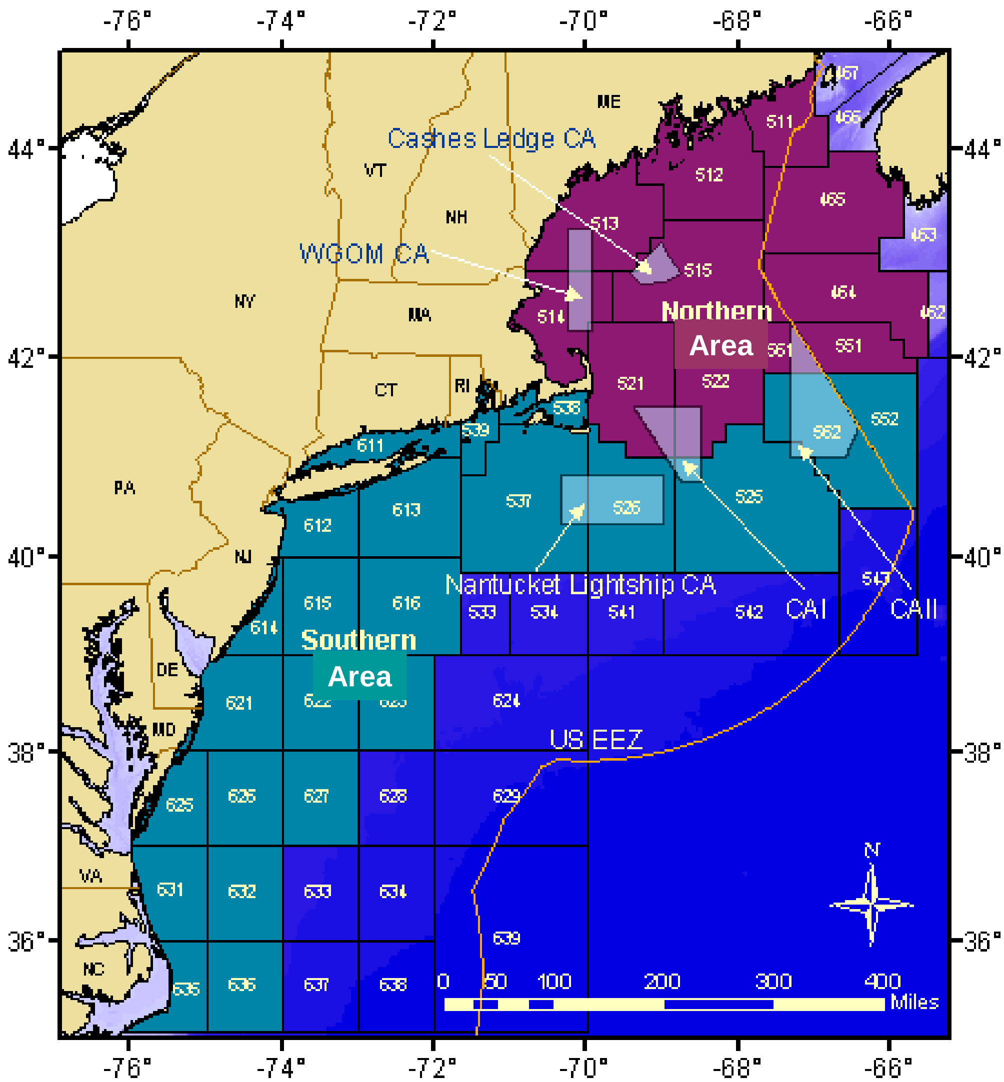


Figure A1. Statistical areas used to define the northern and southern monkfish management areas (from Richards 2006).

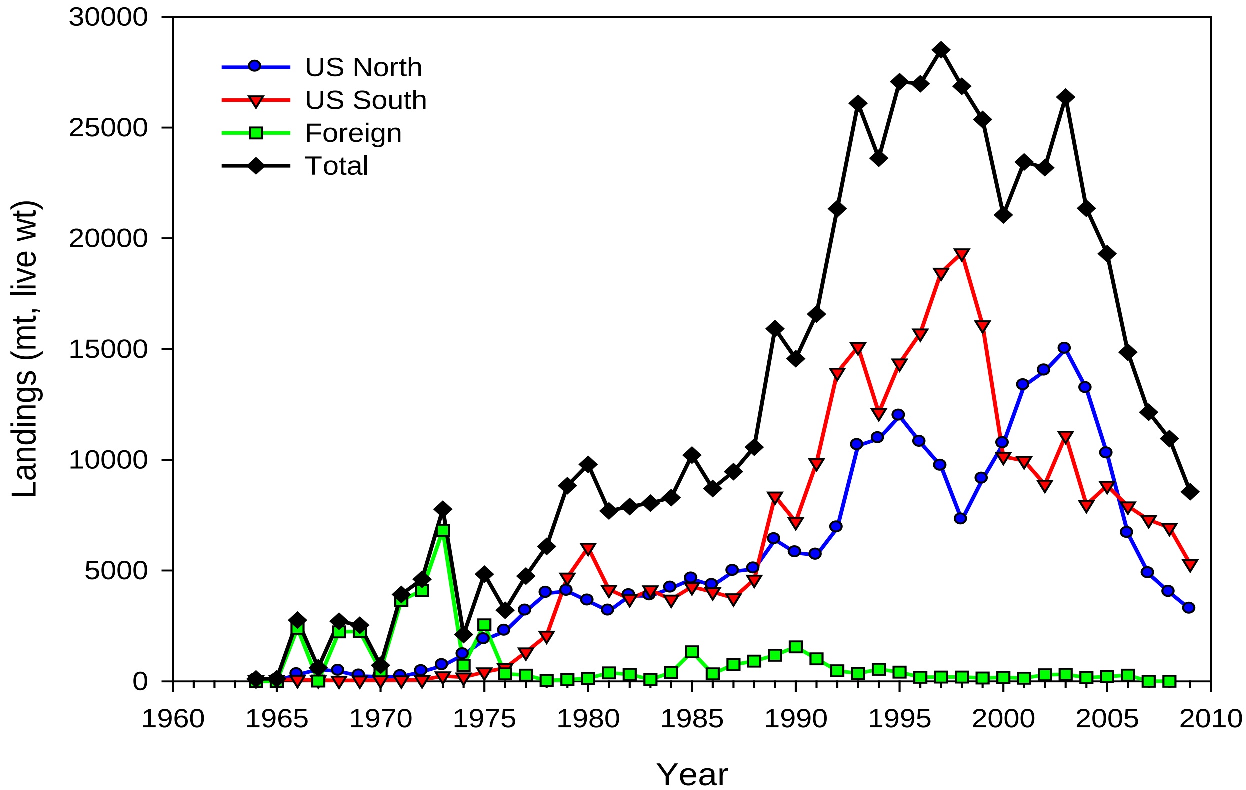


Figure A2. Monkfish landings, by management area and total, 1964-2009.

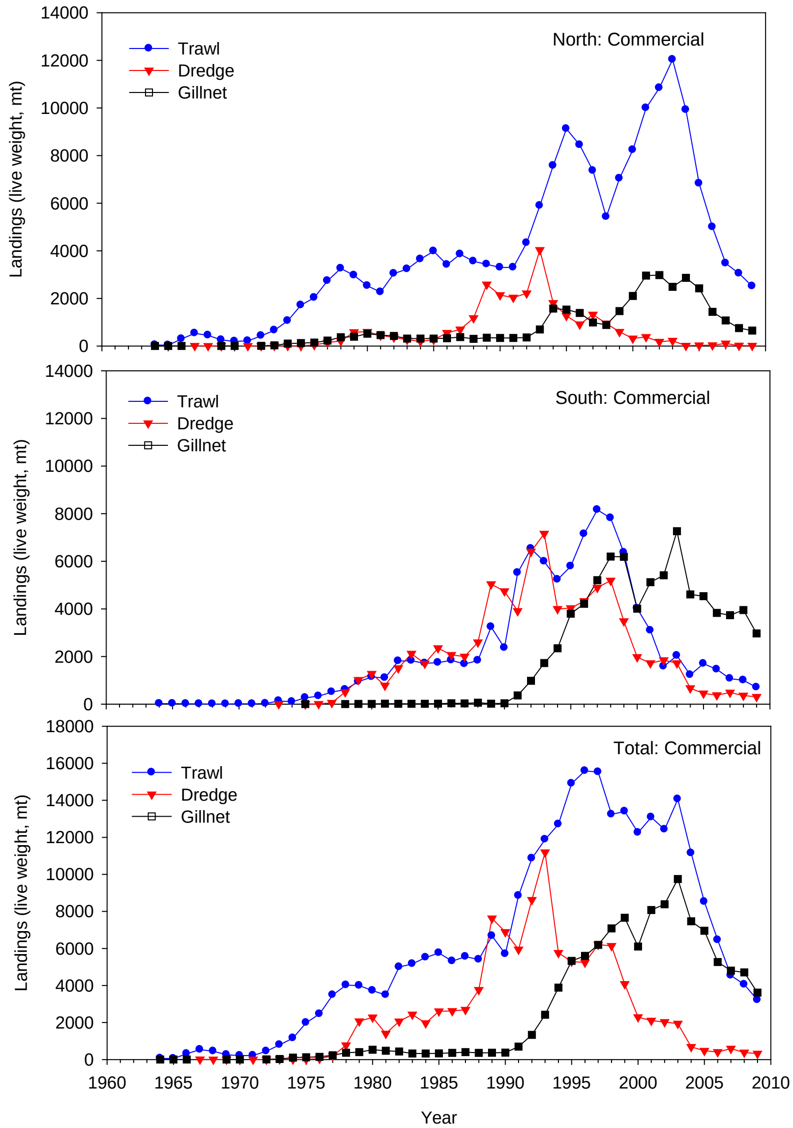
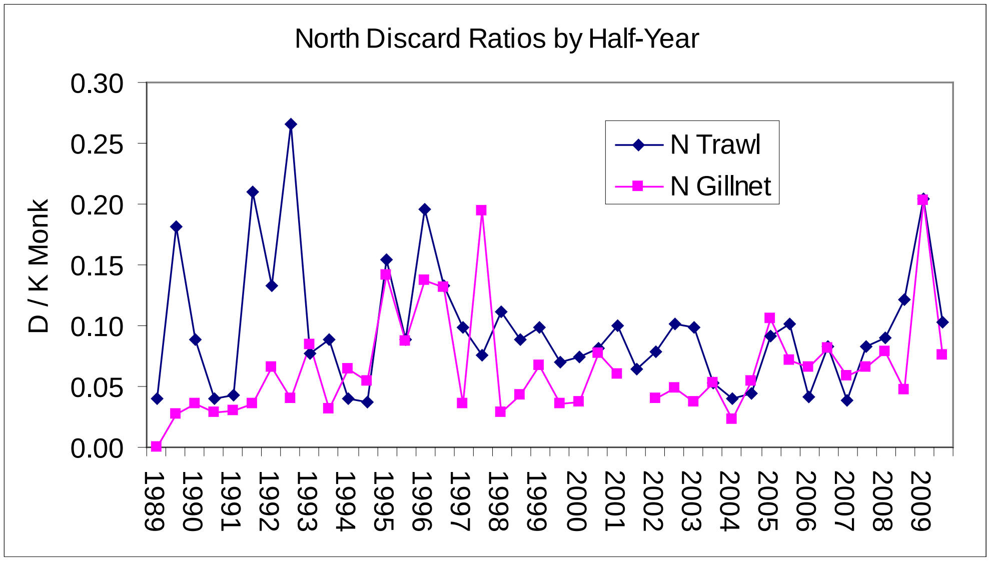


Figure A3. Commercial landings for monkfish by gear type and area.



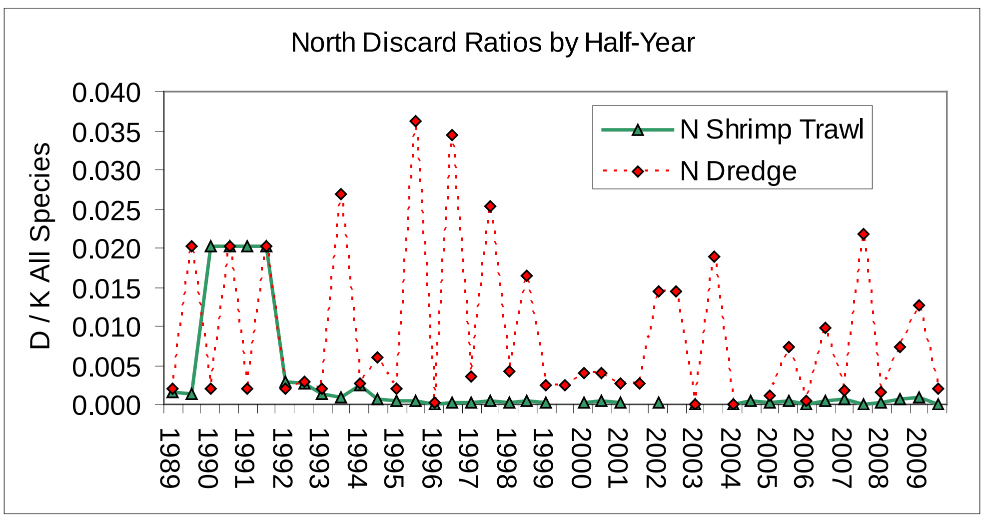


Figure A4. Discard ratios (mt monkfish discarded/mt all species landed) of goosefish by gear and half year using the SBRM methodology in the northern area. Gillnet 2001 half=2 and dredge 2004 half=2 are not shown to preserve scale.

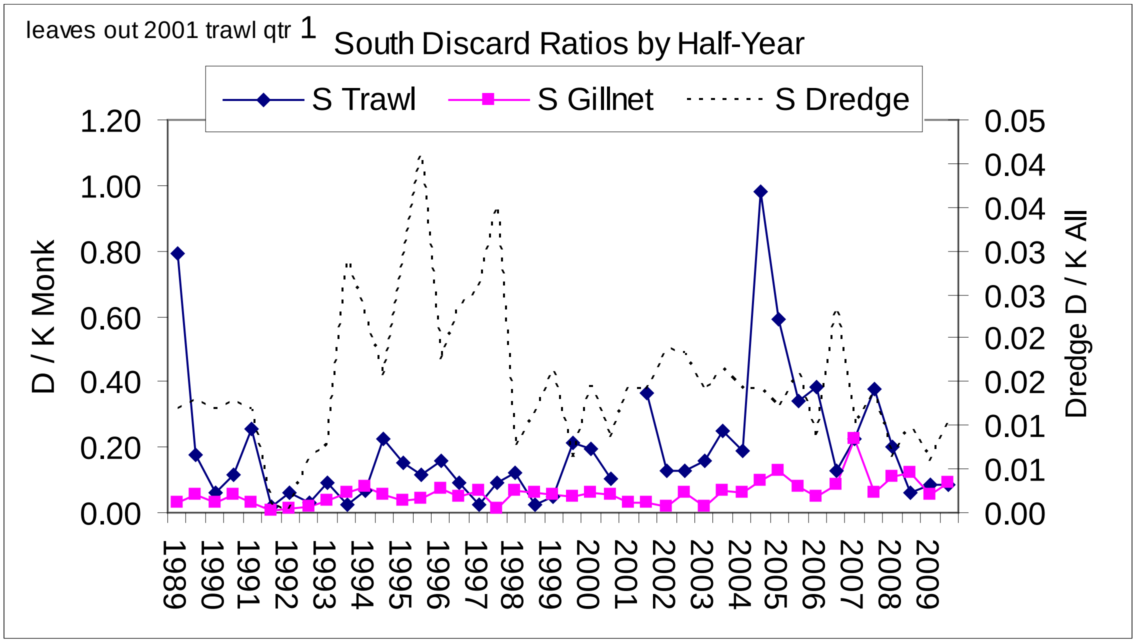


Figure A5. Discard ratios (mt monkfish discarded/mt all species landed) of goosefish by gear and half year using the SBRM methodology in the southern area. Trawl 2001 half=1 not shown to preserve scale.

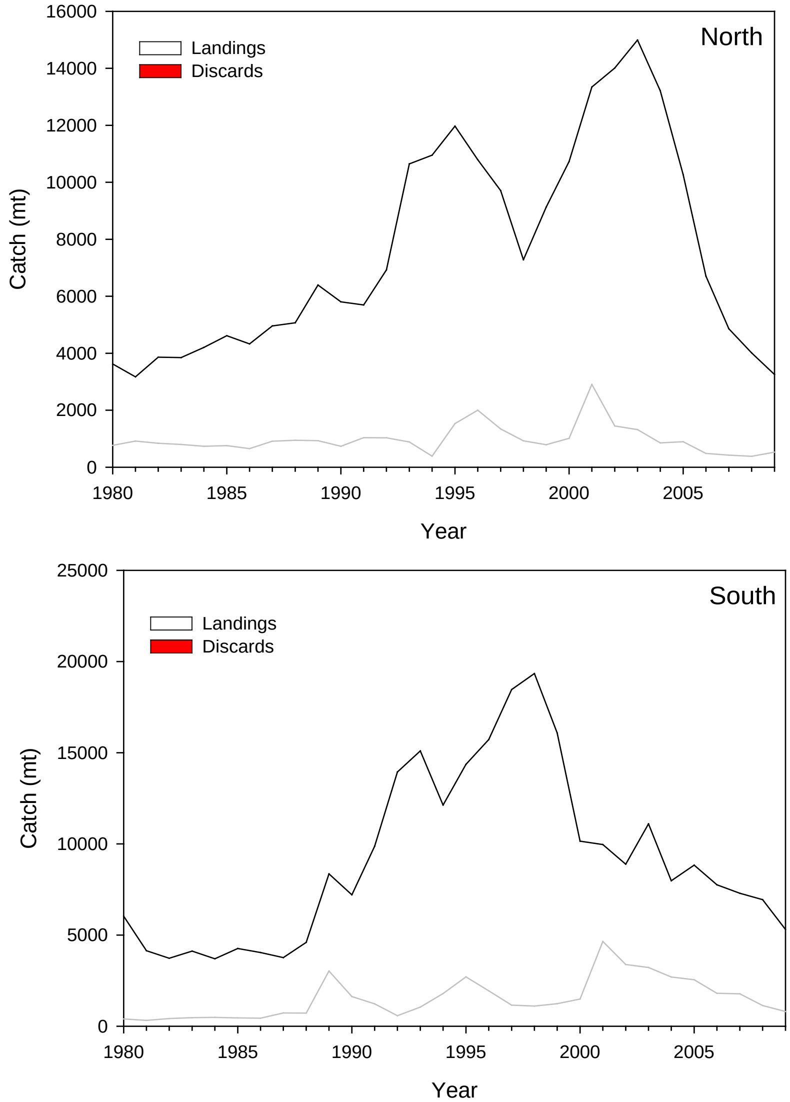


Figure A6. Annual catch of monkfish by management area.

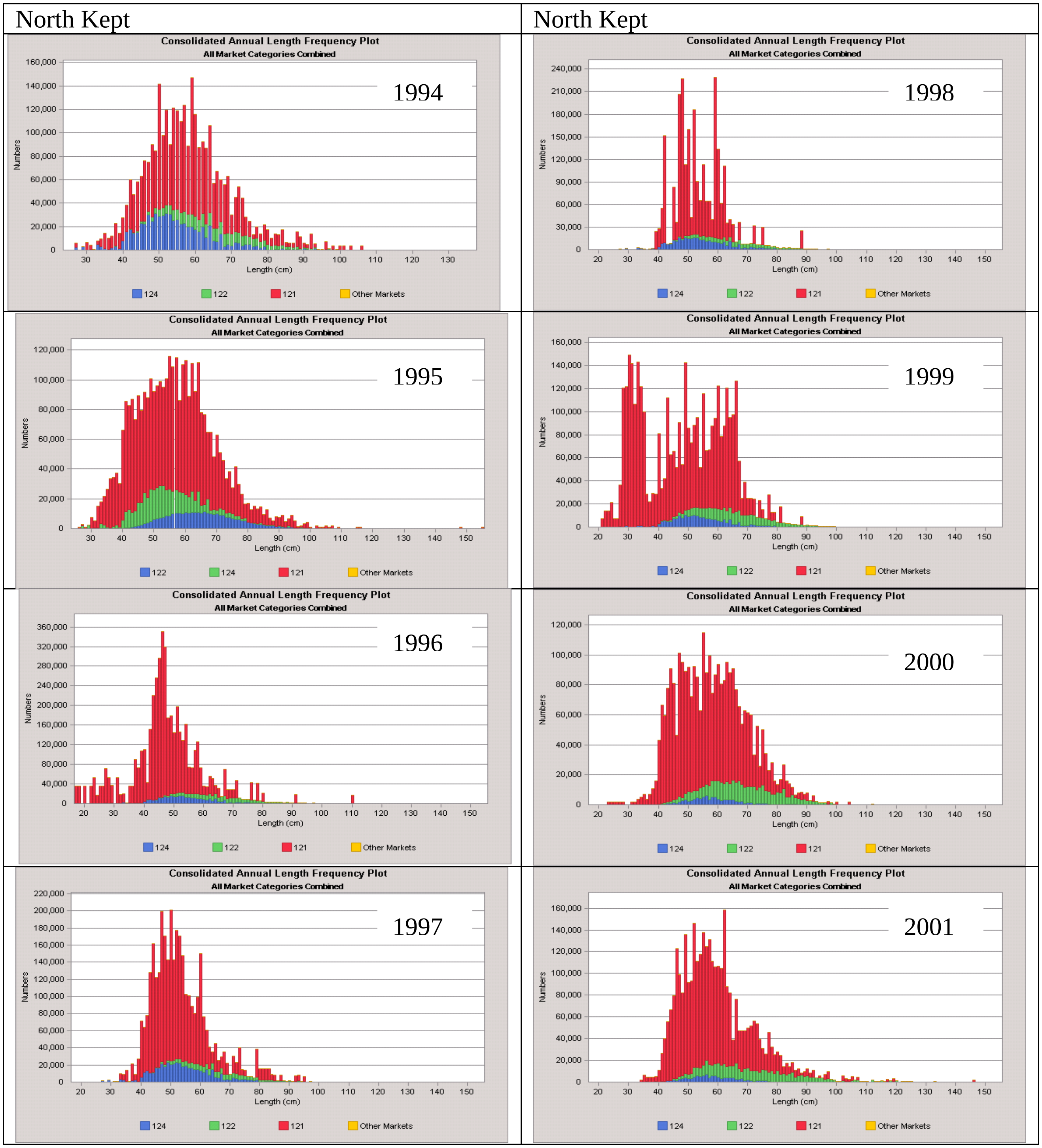


Figure A7. Northern management area, landings at length by gear type, estimated using data from fishery observers. Red=trawls, green=gillnets, blue=dredges, gold=other.

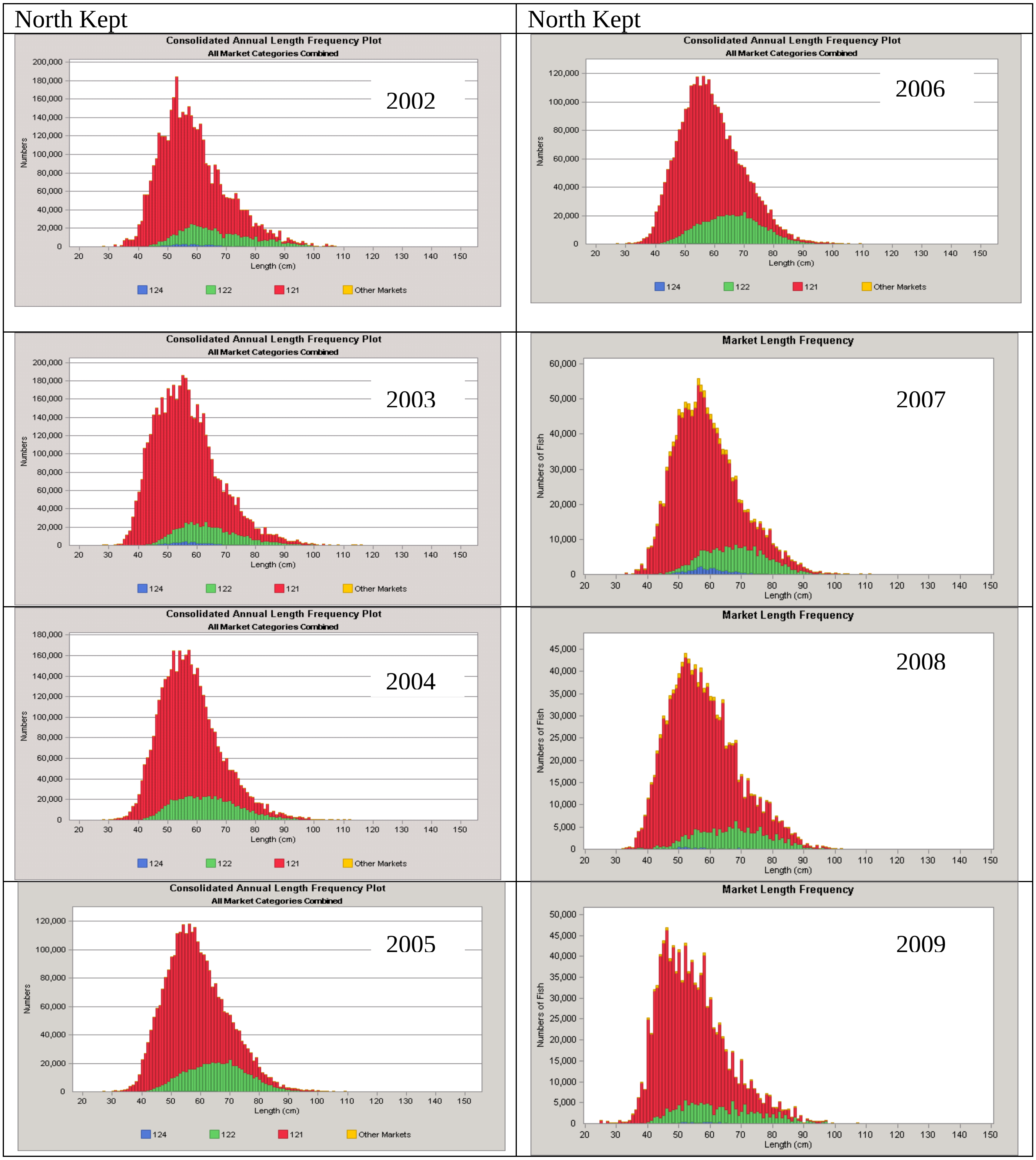


Figure A7, continued. Northern management area, landings at length by gear type, estimated using data from fishery observers. Red=trawls, green=gillnets, blue=dredges, gold=other.

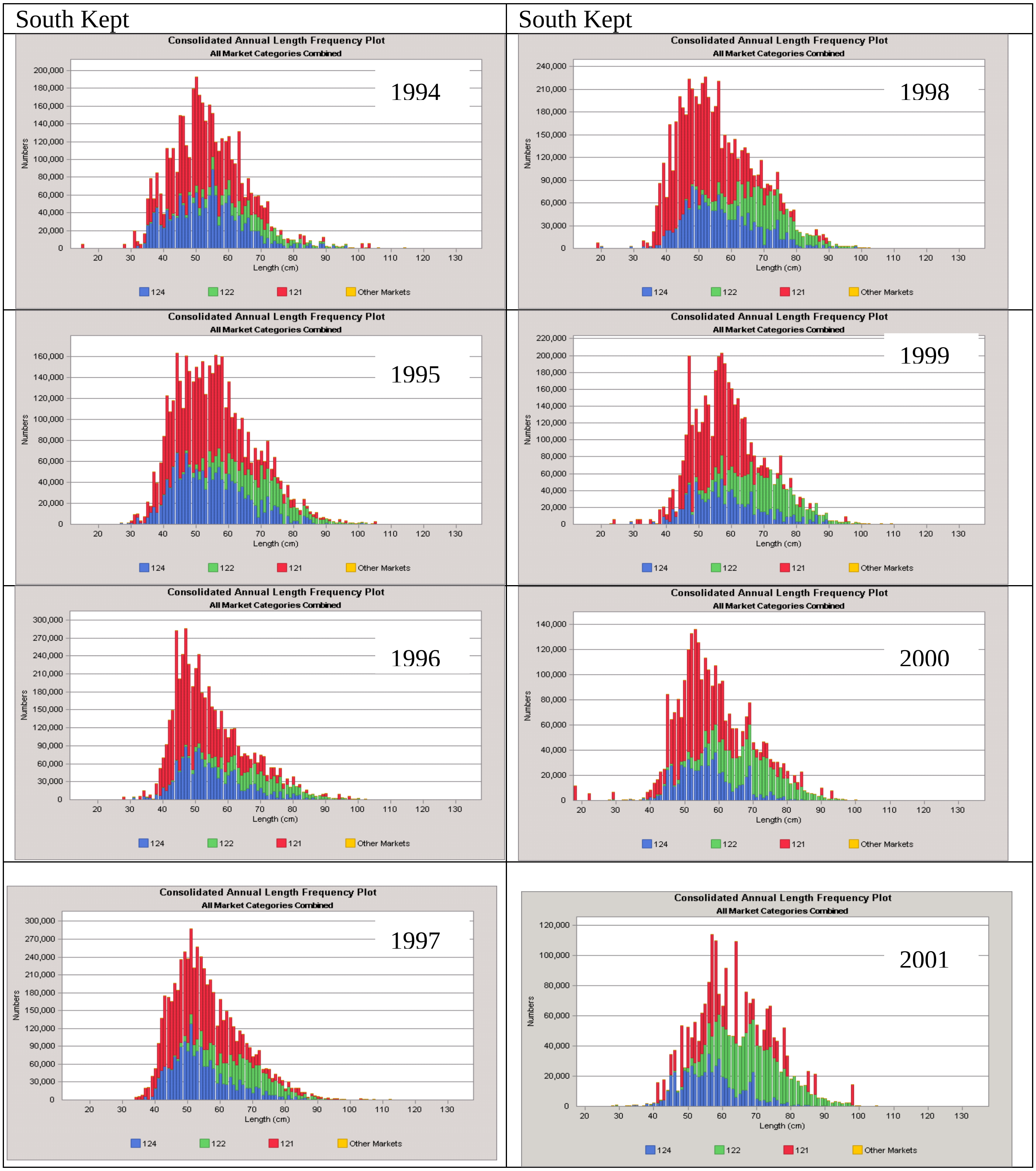


Figure A8. Southern management area, landings at length by gear type, estimated using data from fishery observers. Red=trawls, green=gillnets, blue=dredges, gold=other.

South Kept | South Kept

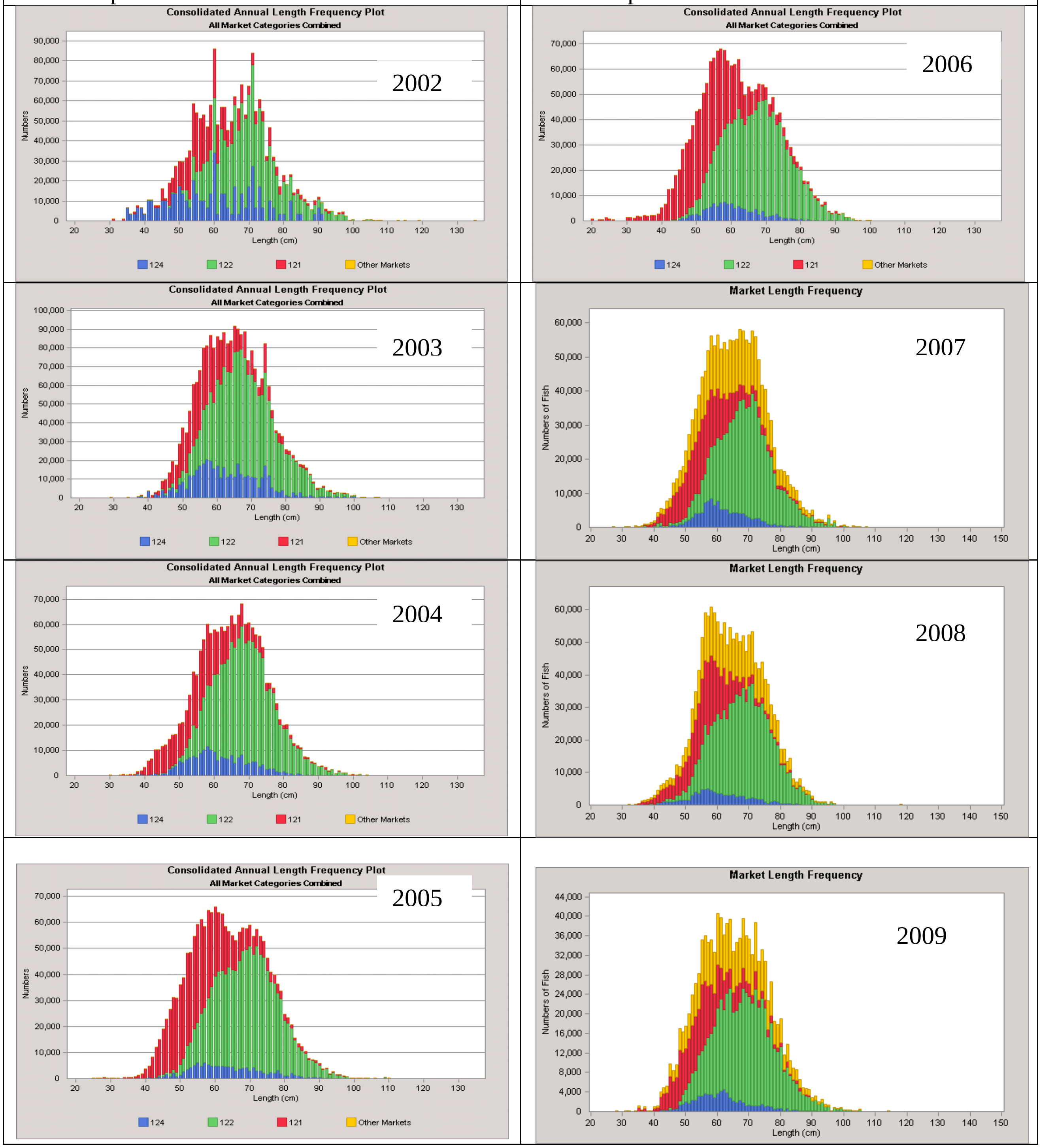


Figure A8, continued. Southern management area, landings at length by gear type, estimated using data from fishery observers. Red=trawls, green=gillnets, blue=dredges, gold=other.

North Discarded	North Discarded

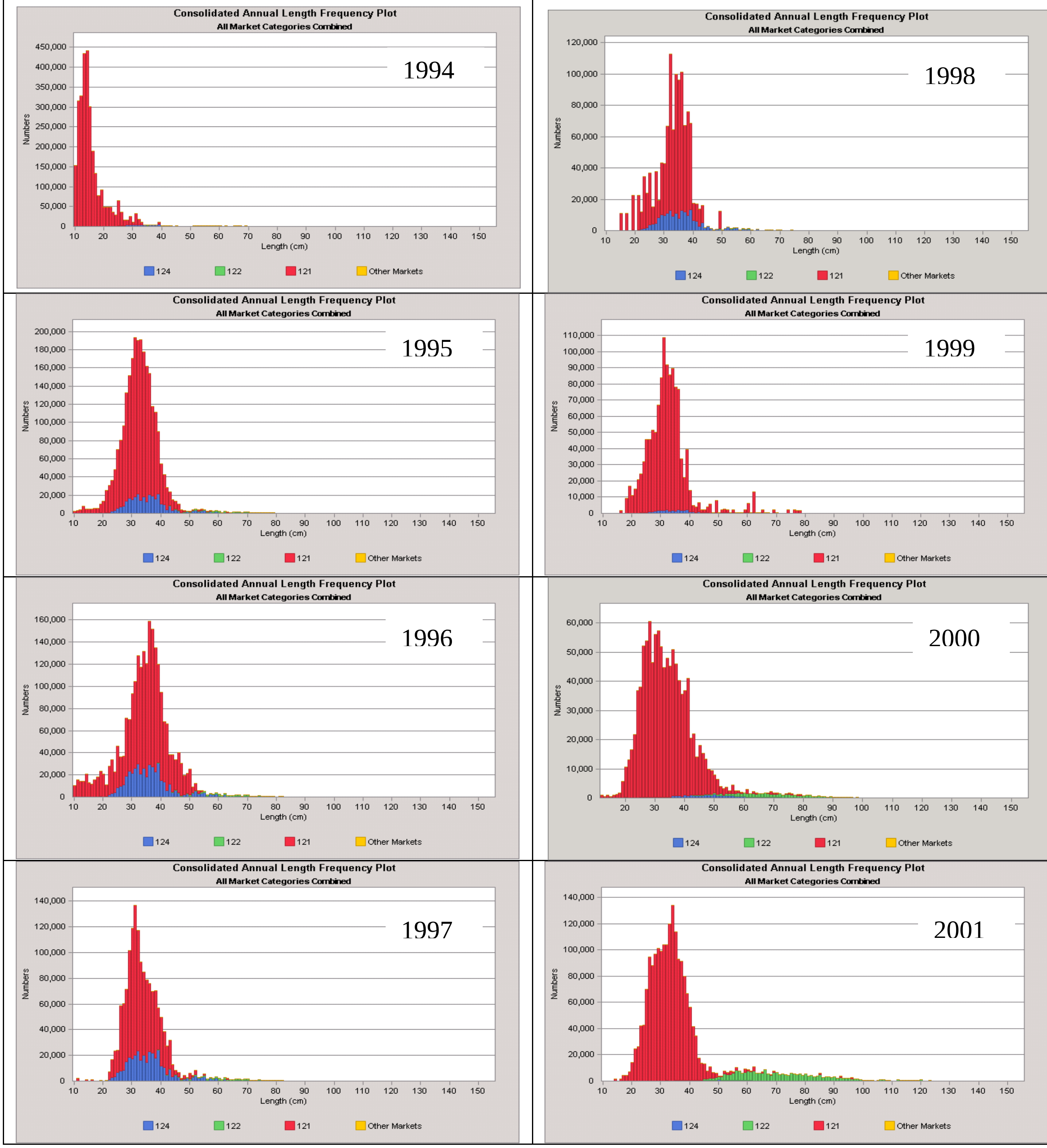


Figure A9. Northern management area, discards at length by gear type, estimated using data from fishery observers. Red=trawls, green=gillnets, blue=dredges, gold=other.

North Discarded | North Discarded

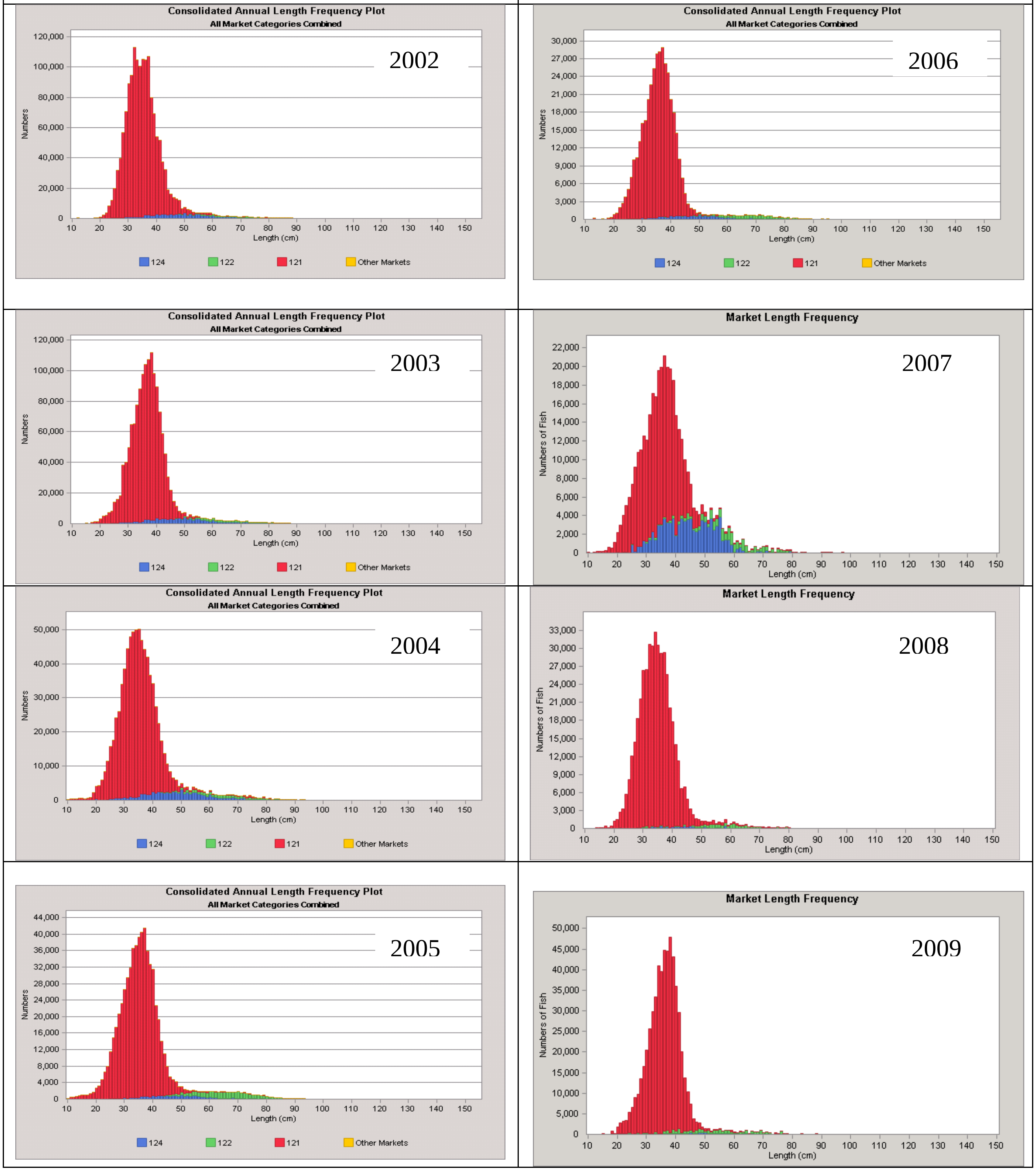


Figure A9, continued. Northern management area, discards at length by gear type, estimated using data from fishery observers. Red=trawls, green=gillnets, blue=dredges, gold=other.

South Discarded	South Discarded

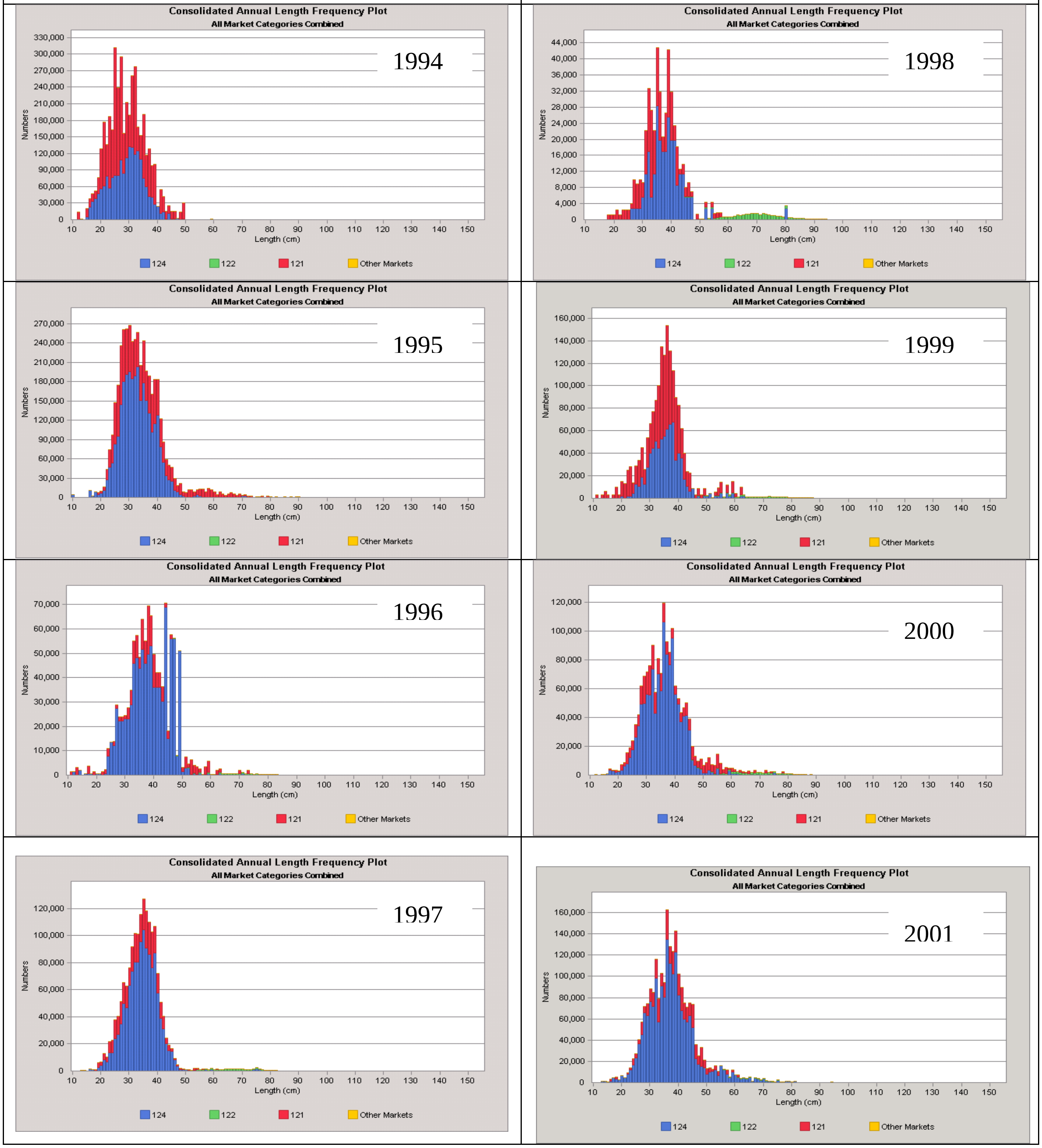


Figure A10. Southern management area, discards at length by gear type, estimated using data from fishery observers. Red=trawls, green=gillnets, blue=dredges, gold=other.

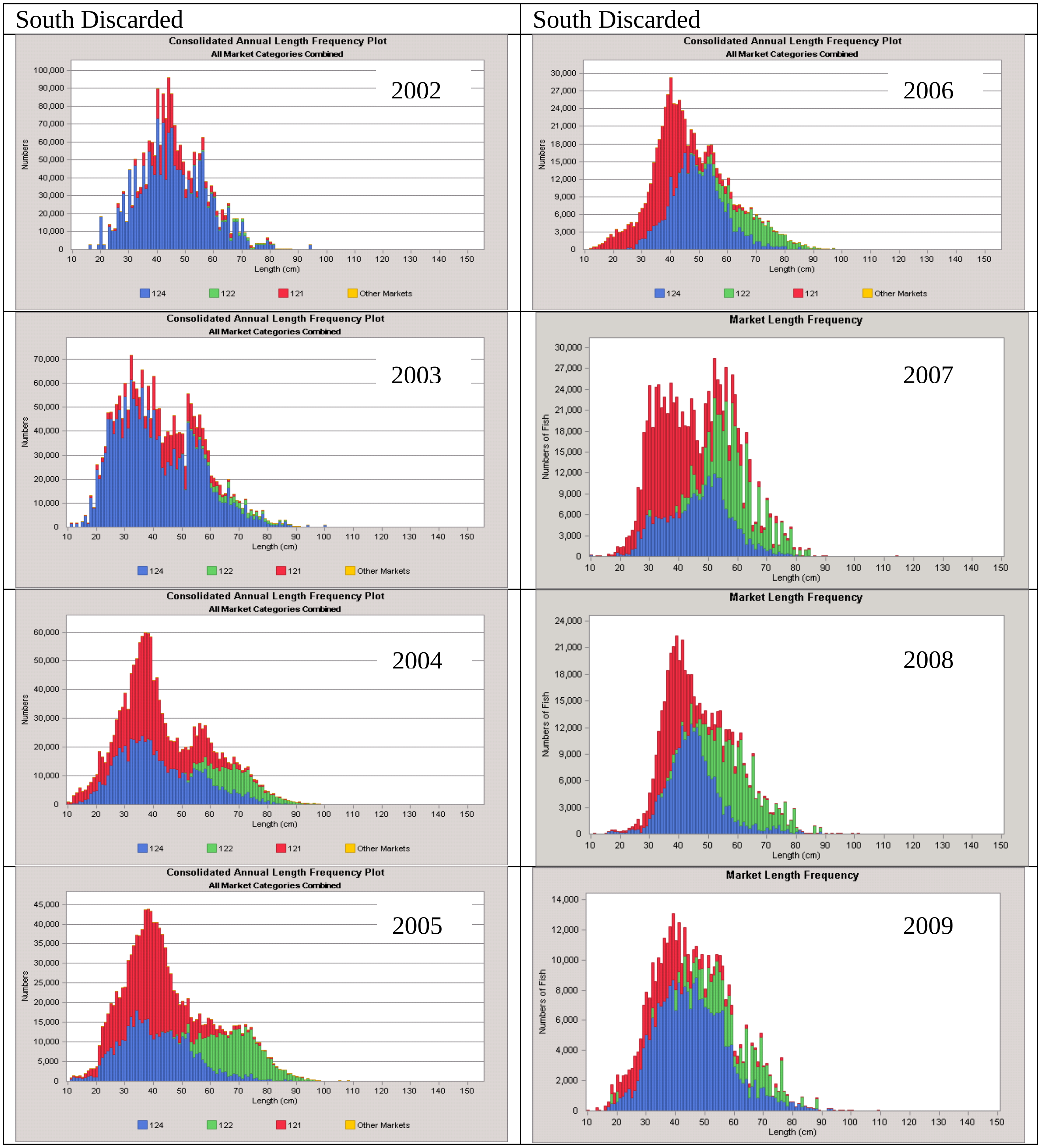


Figure A10, continued. Southern management area, discards at length by gear type, estimated using data from fishery observers. Red=trawls, green=gillnets, blue=dredges, gold=other.

NORTH

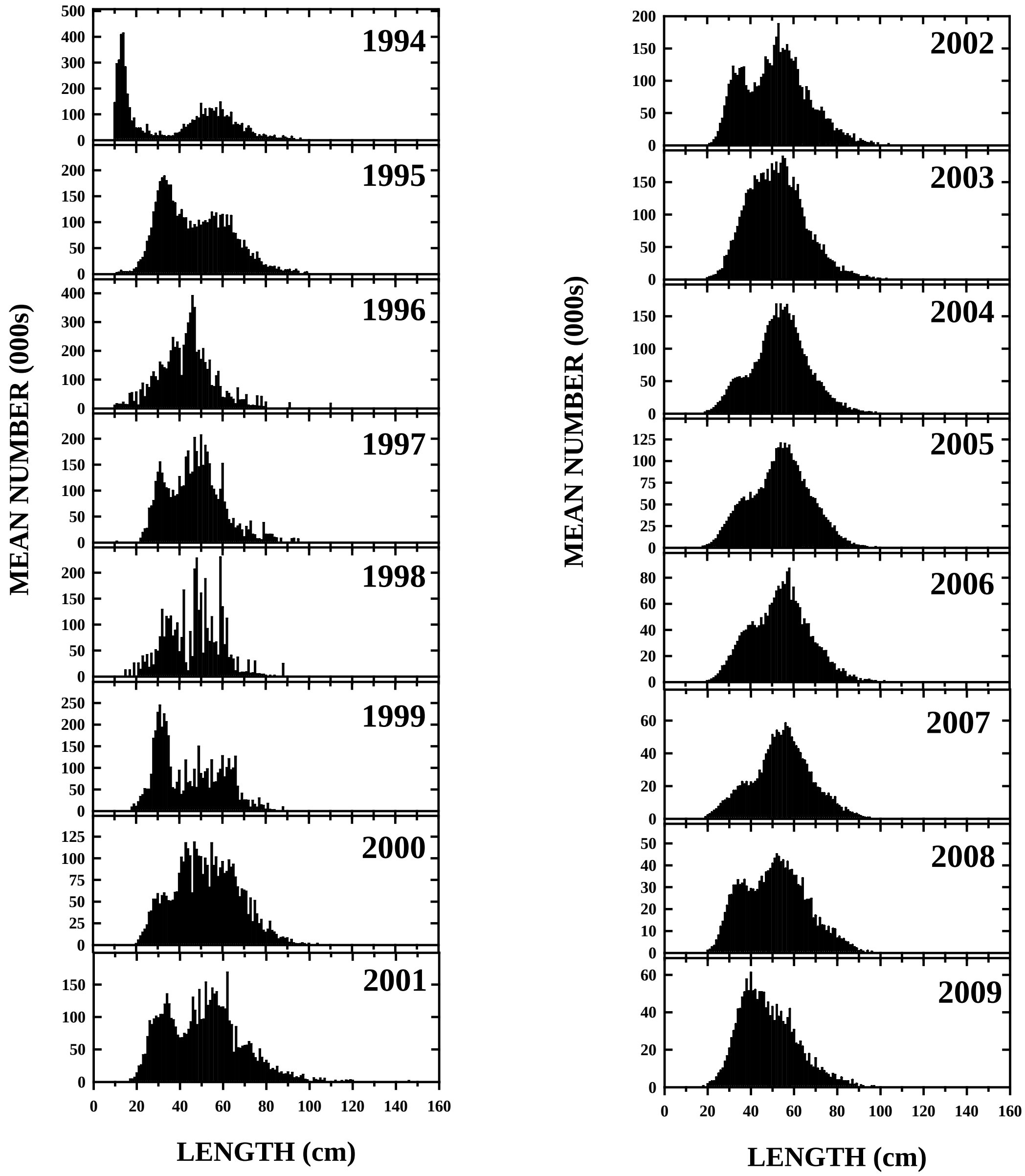


Figure A11. Length composition of commercial catch estimated from observed length samples in the northern management region.

SOUTH

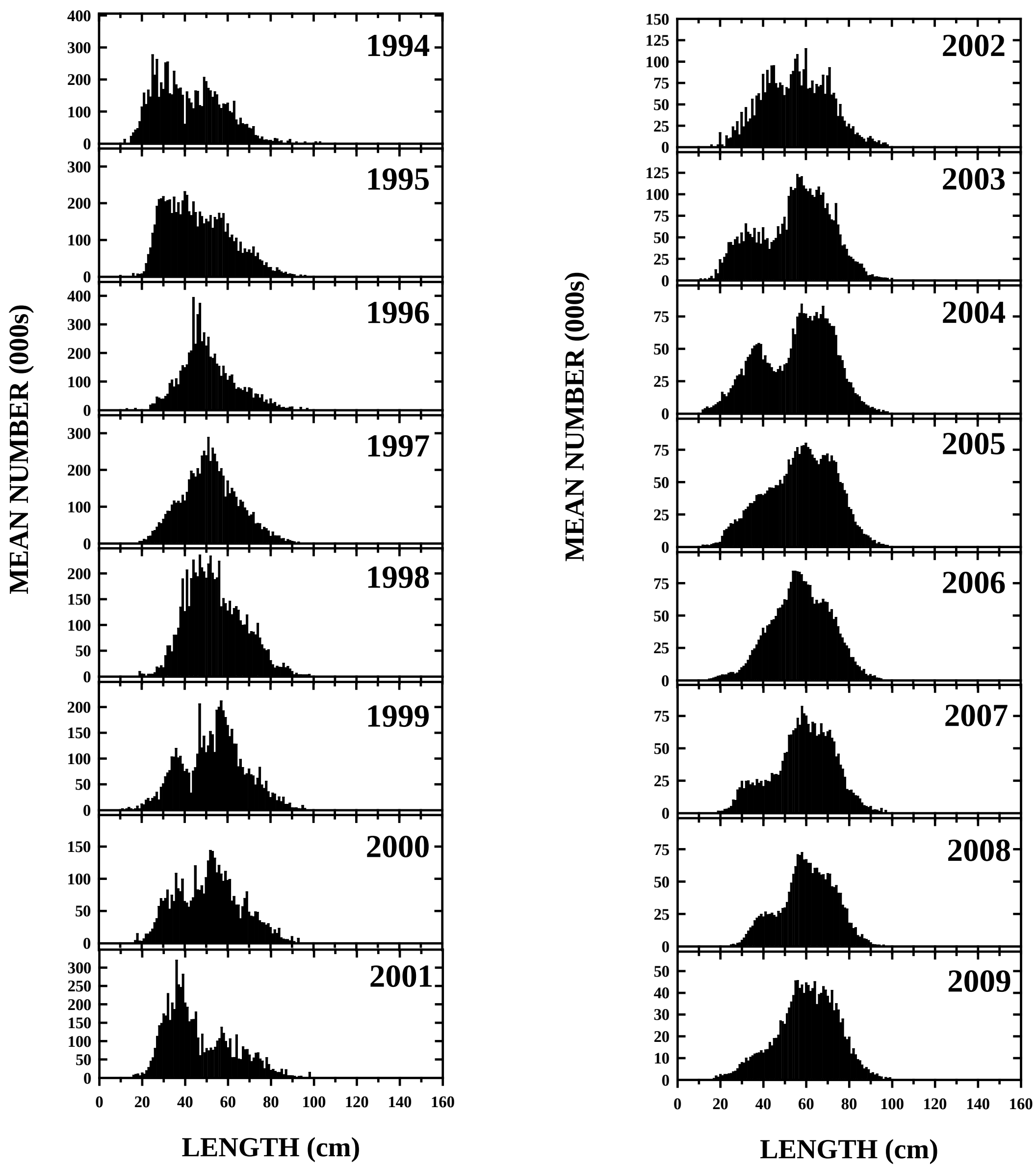


Figure A12. Length composition of commercial catch (discard estimates) estimated from observed length samples in the southern management region.

NORTH + SOUTH

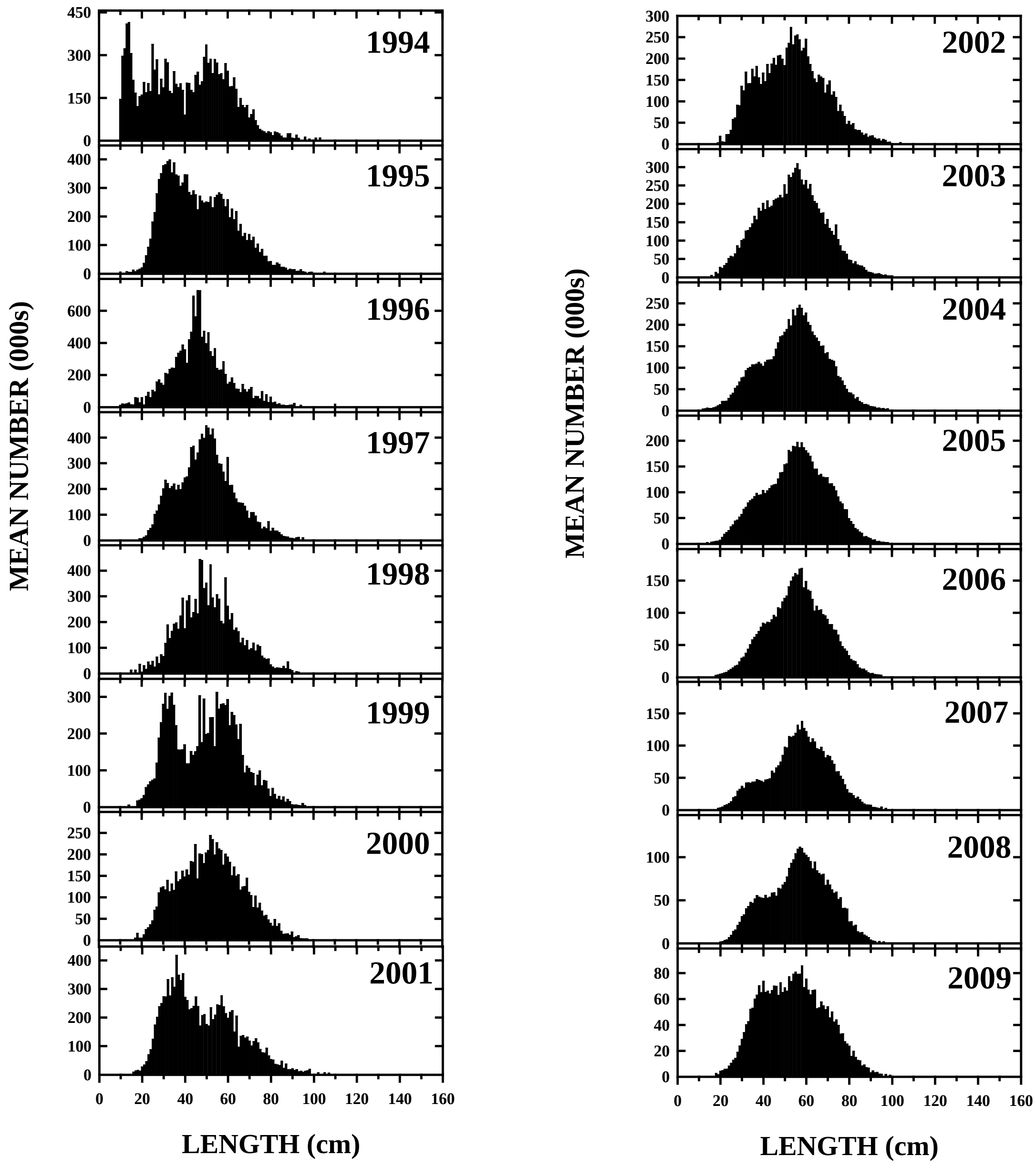


Figure A13. Length composition of commercial catch (discard estimates) estimated from observed length samples in the northern and southern management regions combined.

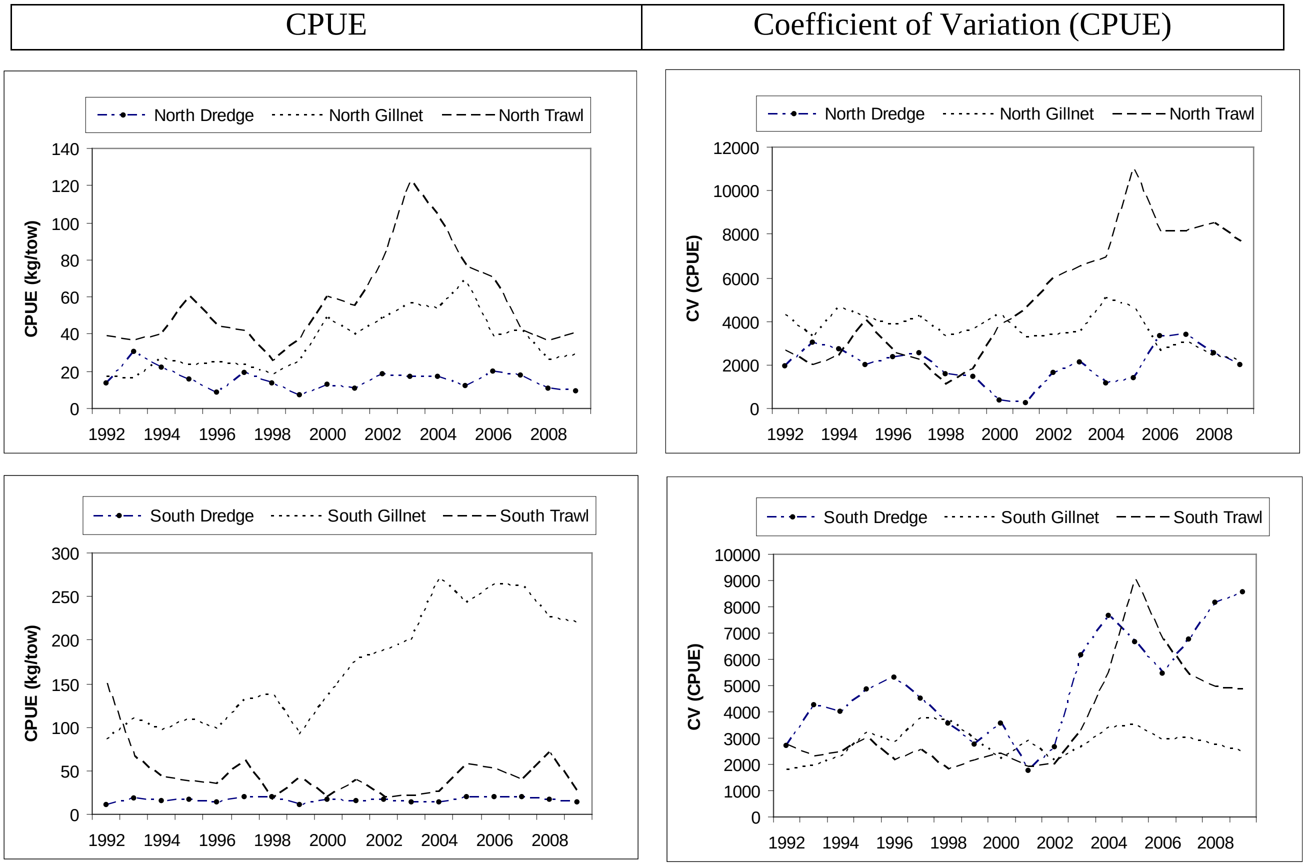


Figure A14. Catch rates of monkfish in the northern and southern management areas from observed tows that caught monkfish by gear-type. Left column, CPUE; right column, coefficient of variation of CPUE estimate.

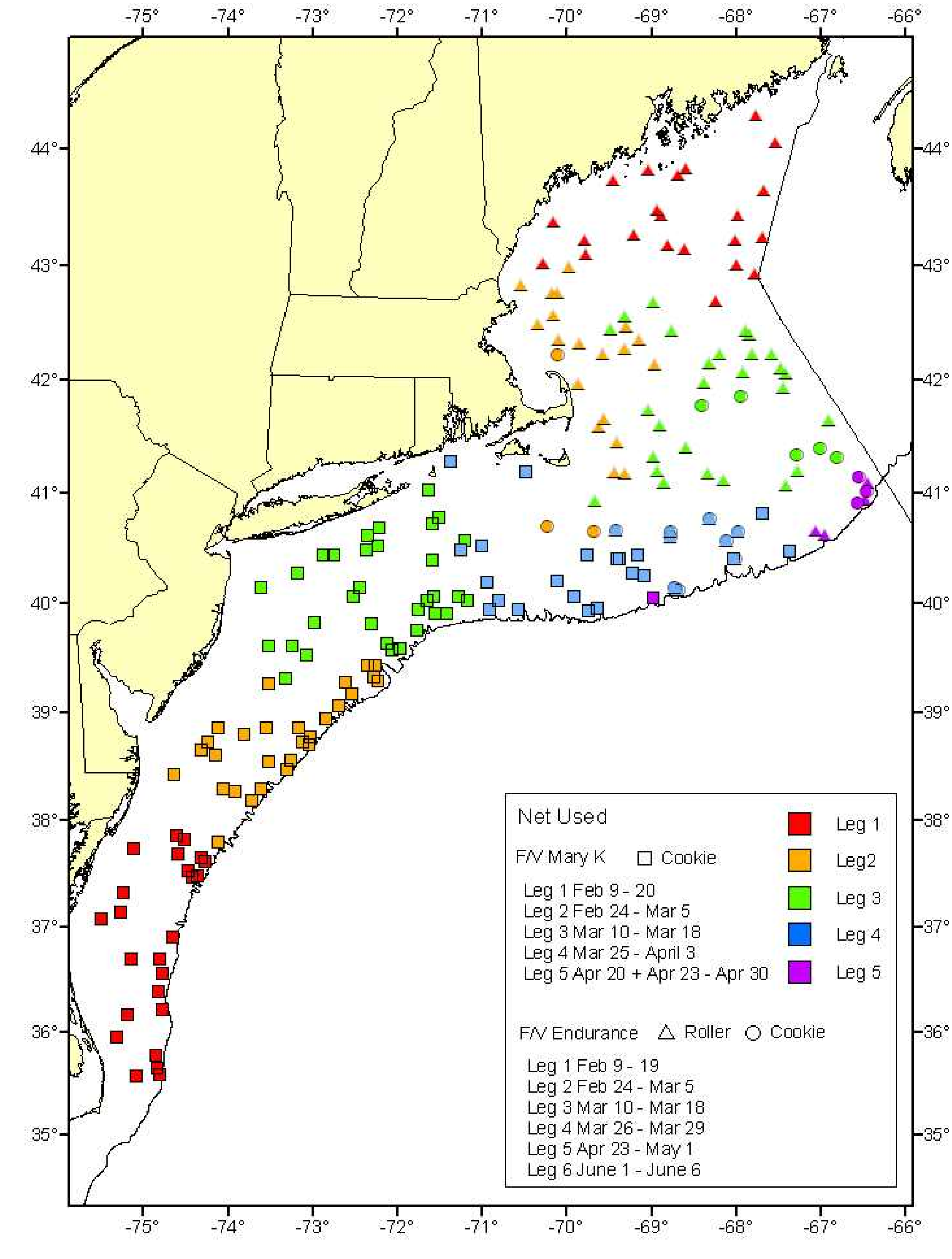


Figure A15. Location of successful survey stations sampled during 2009 cooperative monkfish survey, coded by net type and time of sampling (leg).

2008-2009
408 x 20 cm Fishing circle Trawl with 7"cookie sweep.
Design spread at end of bottom wing web 36 meters for a wing end angle of 24°.
Headrope 49 meters. Sweep 56 meters.
Ground gear 82 feet. End of ground gear to back straps' door attachments 36 feet.
Sensor to back straps' door attachments 6 feet.

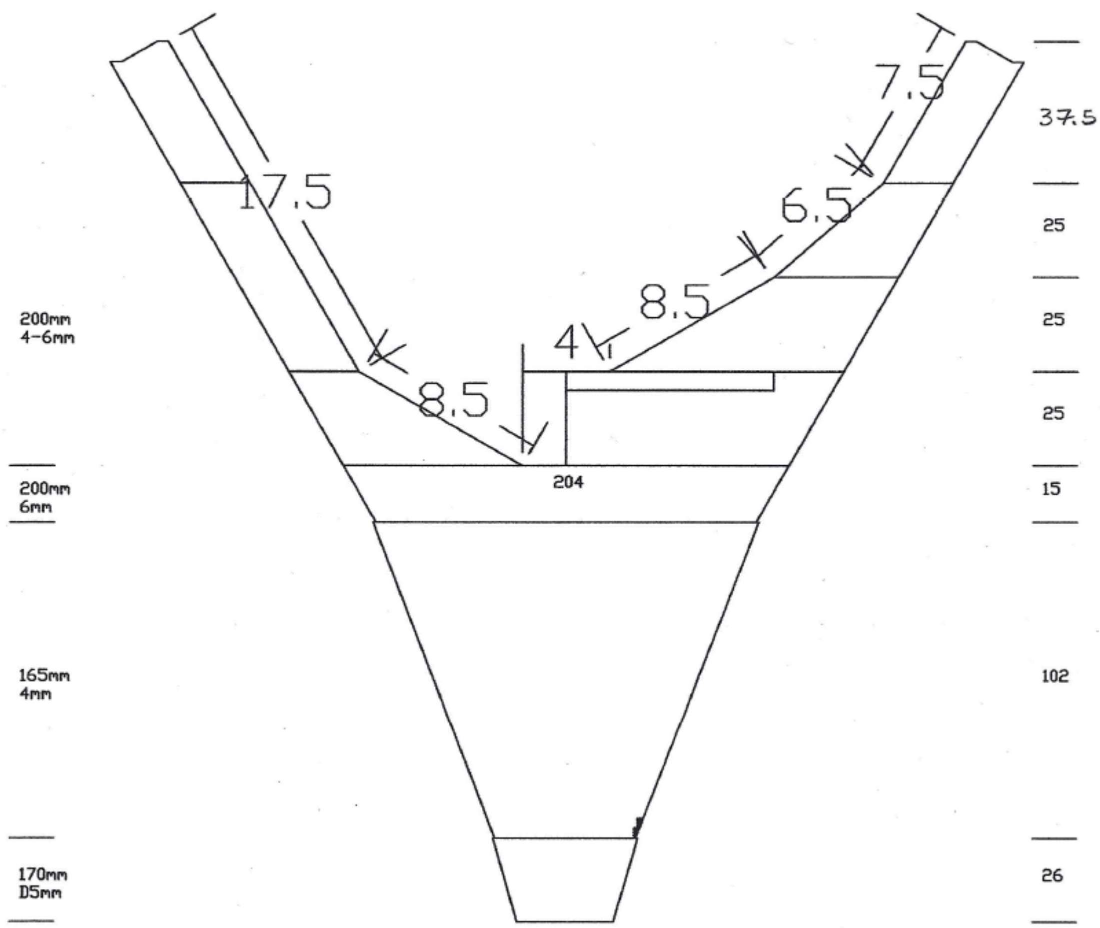


Figure A16. Plan for flat net used on F/V Endurance during 2009 cooperative survey.

F/V ENDURANCE
2008-2009
508 x 20 cm Fishing circle Trawl with rock hopper sweep.
Design spread 45.6 meters for a wing end angle of 23°.
Headrope 79 meters.
Sweep 86 meters (10.4 meters w/o web).
Center section 40 meters w/ 14" discs. Wing sections 22.7 meters w/ 12" discs. 30cm chain.
Ground gear 60 feet. End of ground gear to back straps' door attachment 36 feet.
Sensor to back straps' door attachment 6 feet.

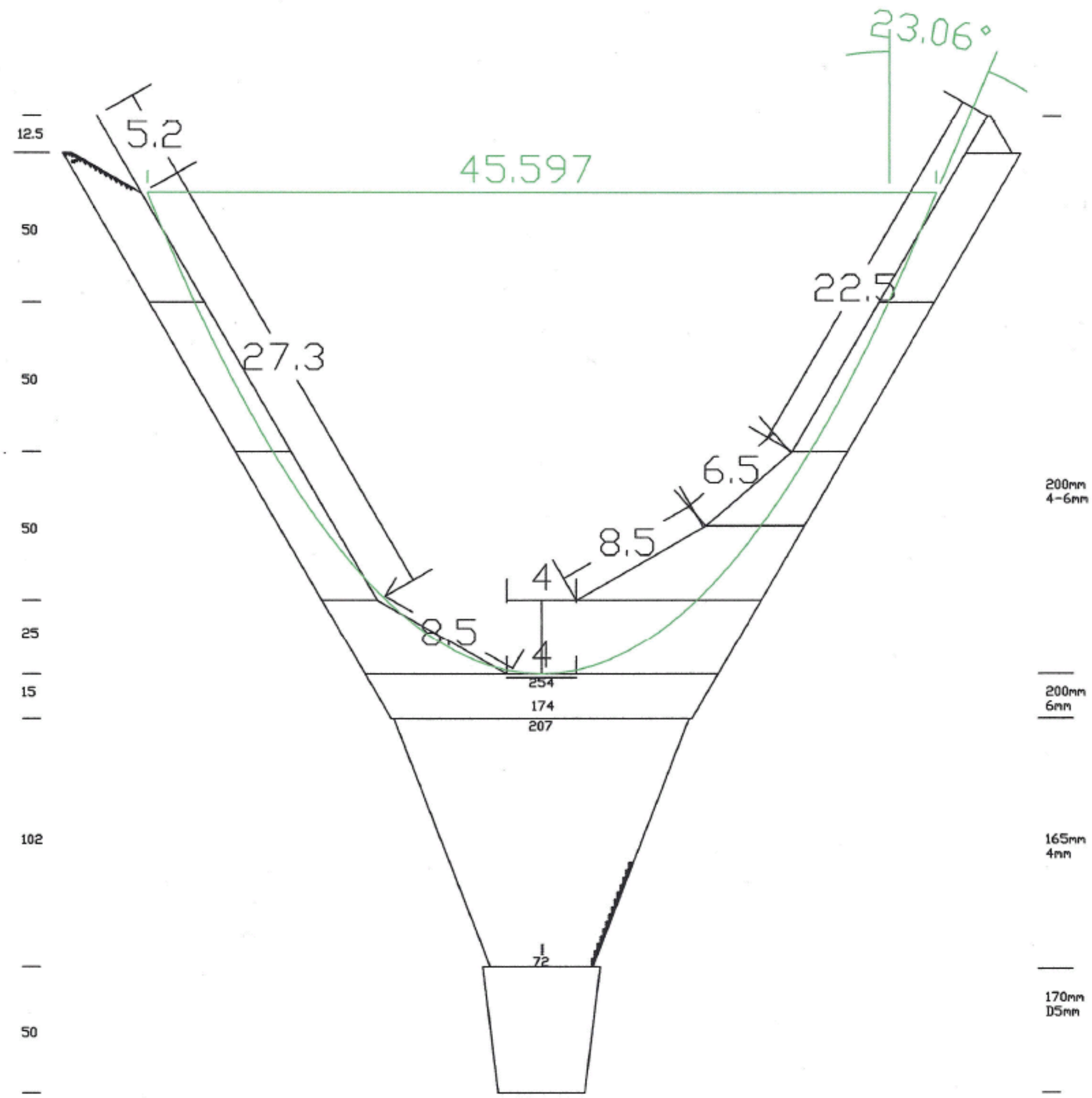


Figure A17. Plan for rockhopper net used on F/V Endurance during 2009 cooperative survey.

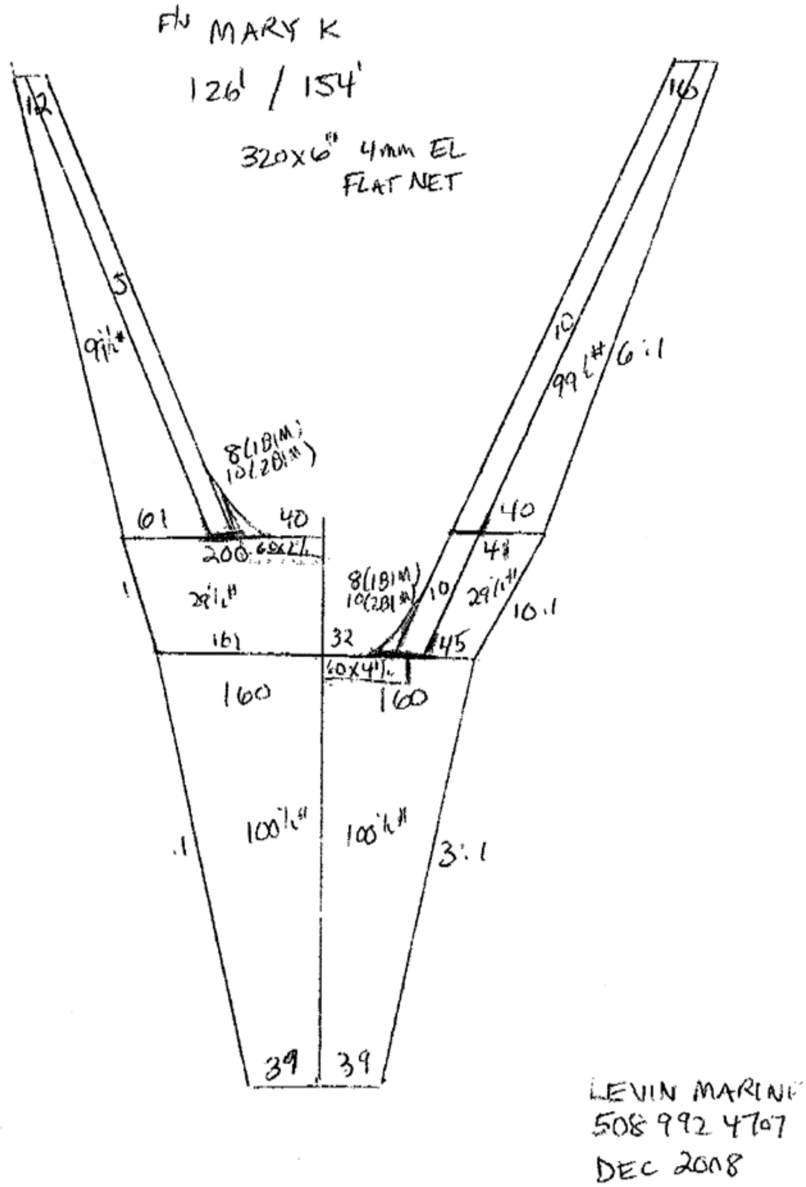
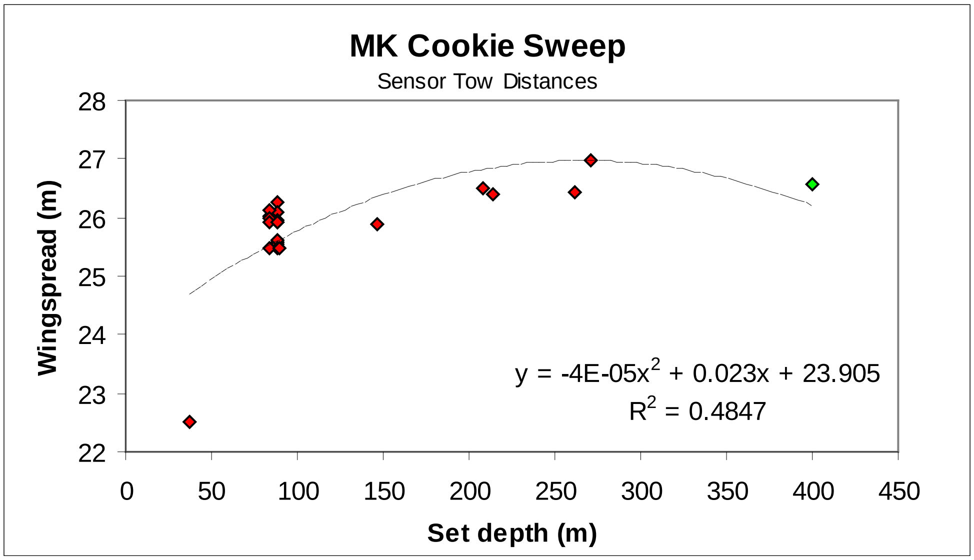


Figure A18. Plan for net used on F/V Mary K during 2009 cooperative survey.



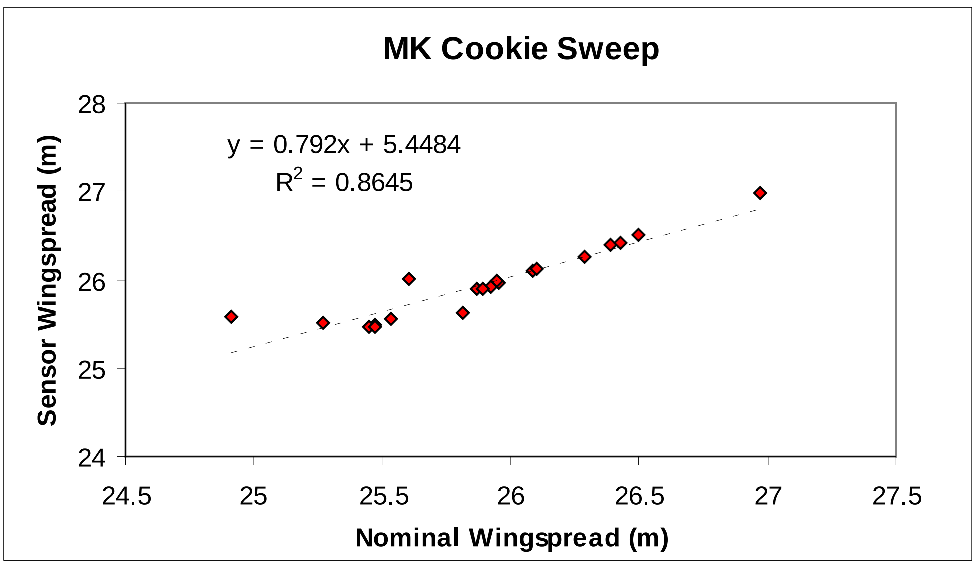


Figure A19. (A) Relationship between depth and wingspread for the cookie sweep net used on the Mary K, 2009 cooperative survey. Data are from mensuration tows and depletion experiments with good quality bottom contact and wingspread measurements, trimmed to sensor tow length before averaging for each tow. Point at 400 m is average wingspread for tows > 200 m set depth, not an observed value; maximum depth with observed wingspread was 271 m. Point at 37 m is based on only 6 wingspread readings. (B) relationship between average wingspread during nominal tow vs. sensor tow duration.

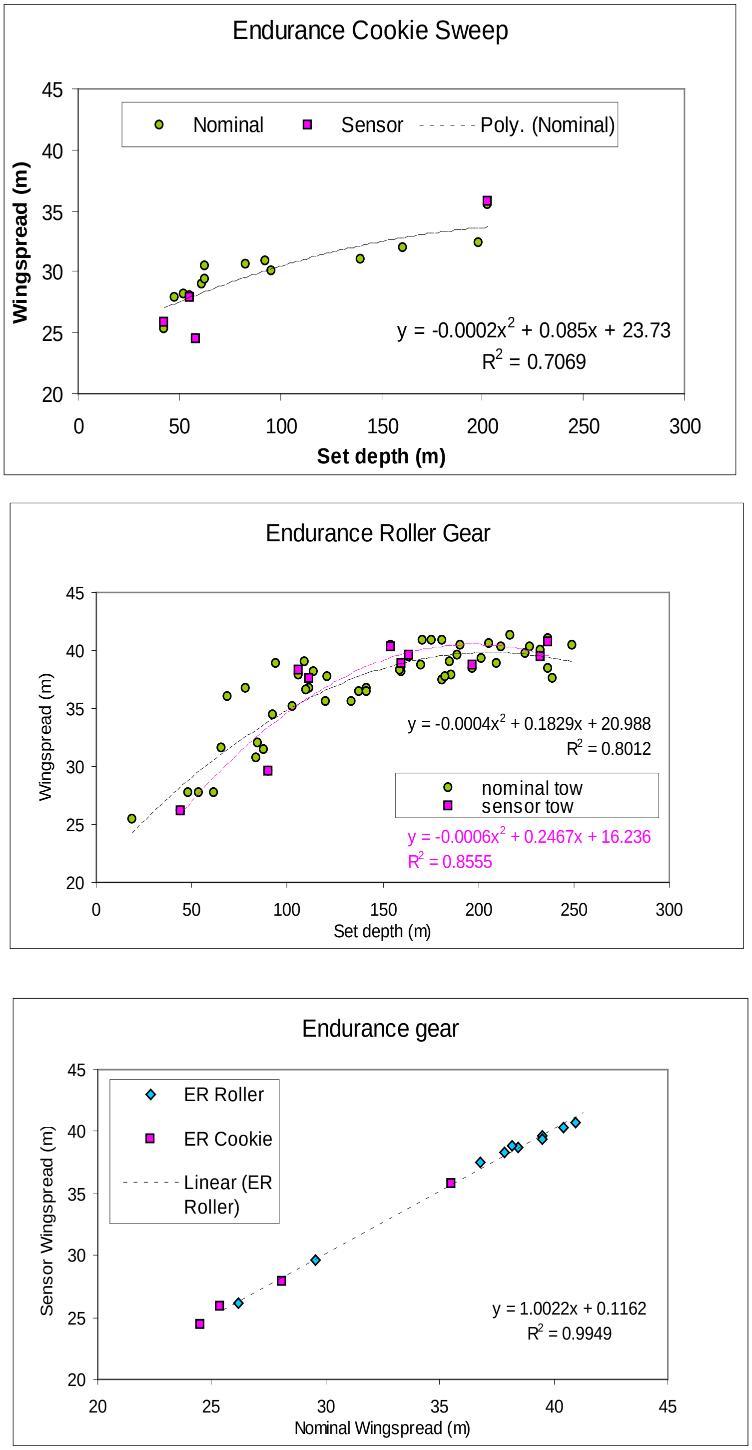
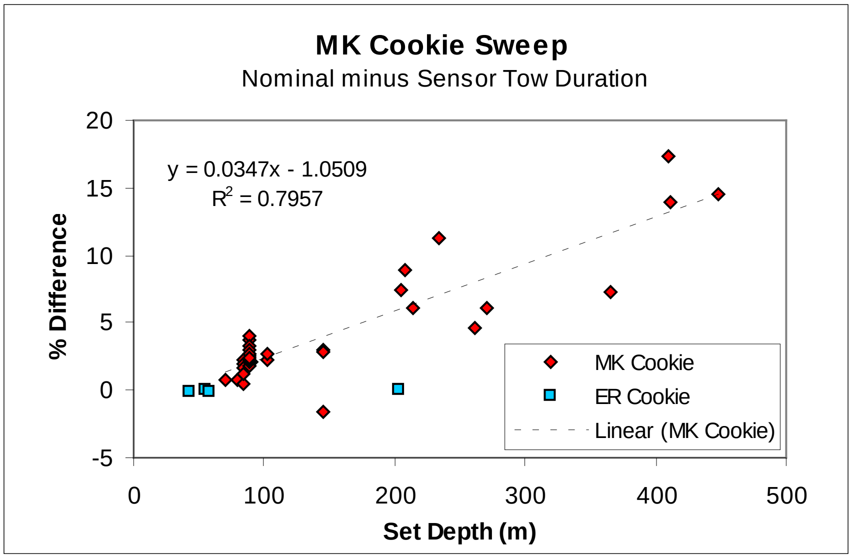


Figure A20. Wingspread-depth relationship for Endurance (A) cookie sweep net, (B) roller gear net, and (C) relationship between average wingspread during nominal tow vs. sensor tow duration.



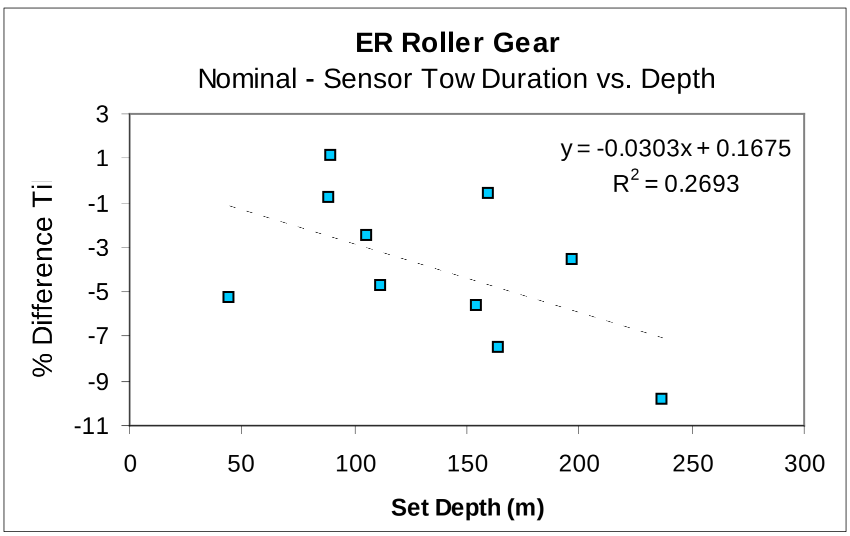


Figure A21. Relative difference between tow duration estimated from sensor data (tows with good bottom contact readings) and nominal tow duration for each net. (A) Endurance and Mary K cookie sweeps, (B) Endurance roller sweep.

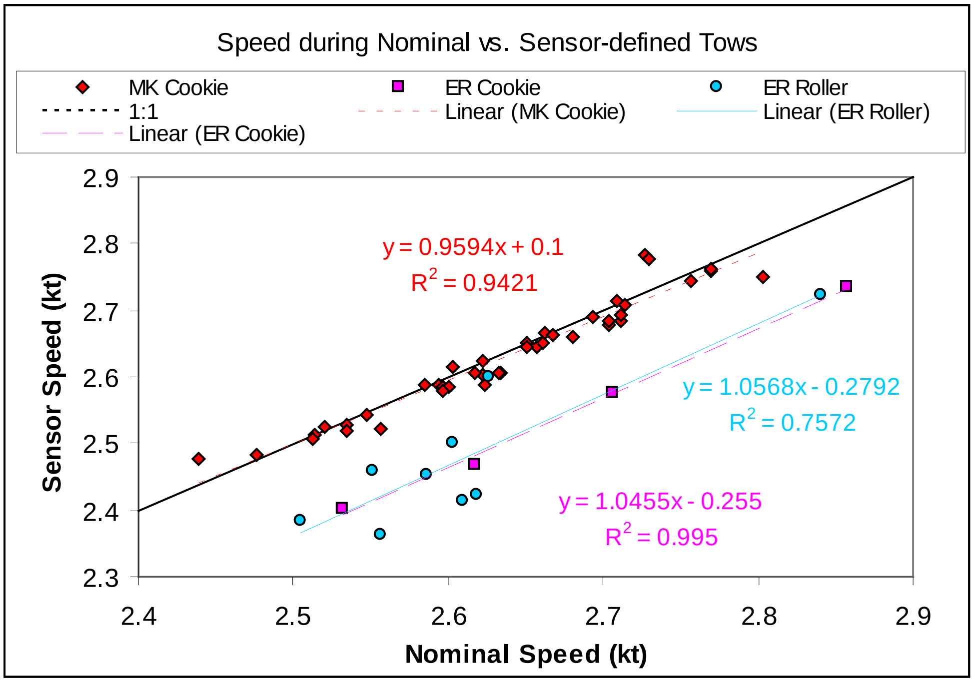


Figure A22. Adjustments to nominal average speed for tows with no bottom contact sensor data to define sensor tow length (and average sensor tow speed).

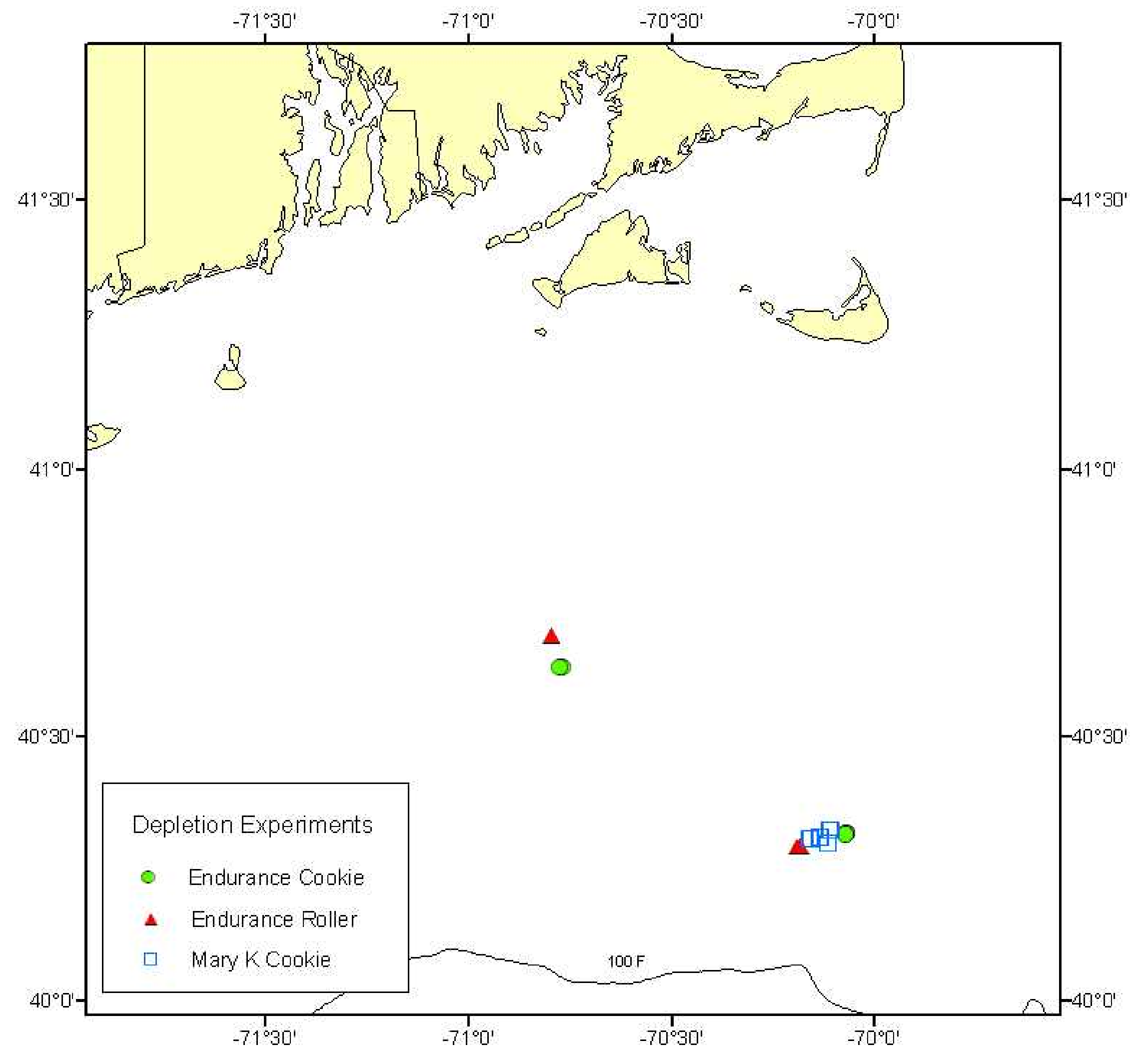


Figure A23. Location of depletion experiments for the 3 net types used in the 2009 cooperative monkfish survey.

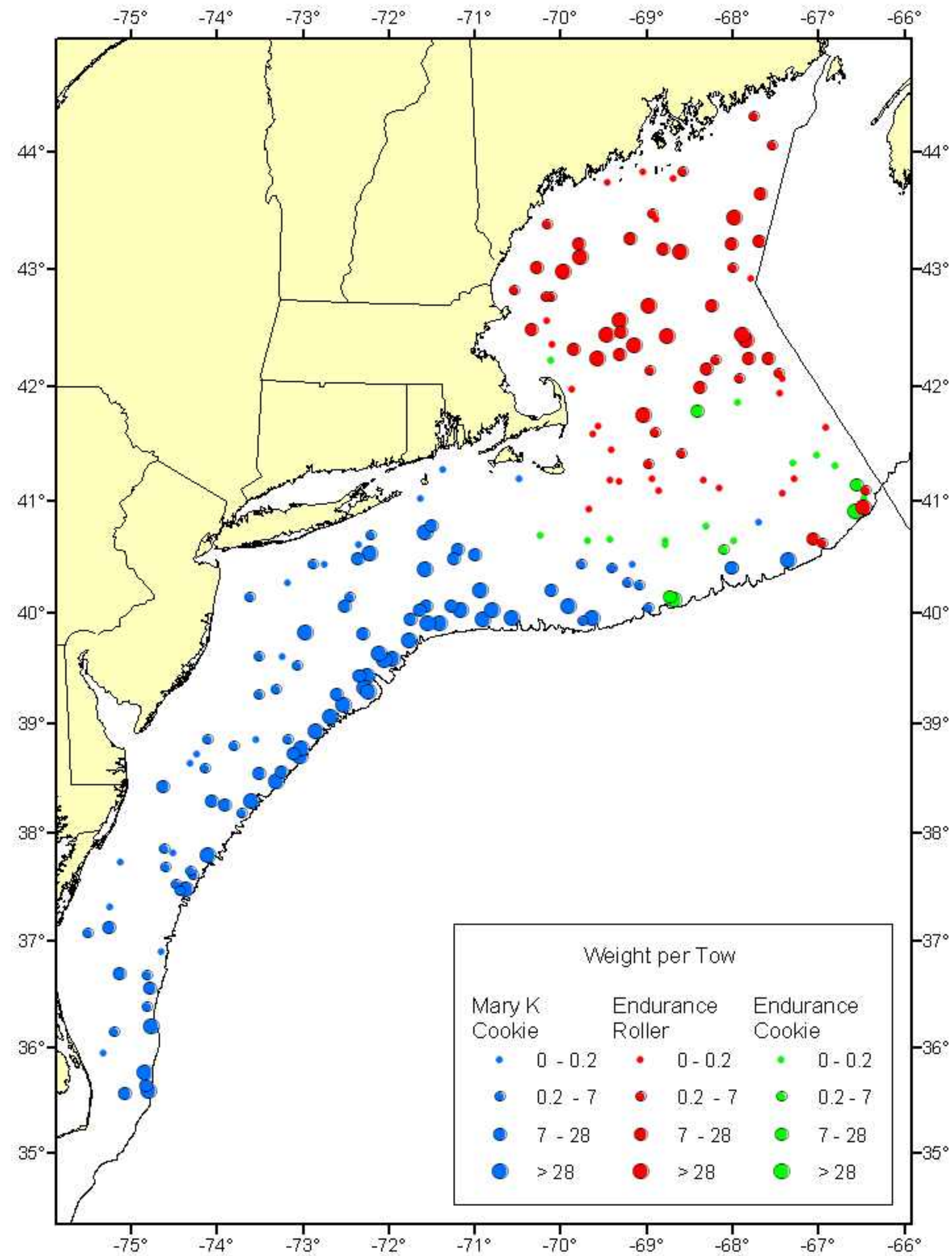


Figure A24. Nominal catch per tow (kg) coded by net type.

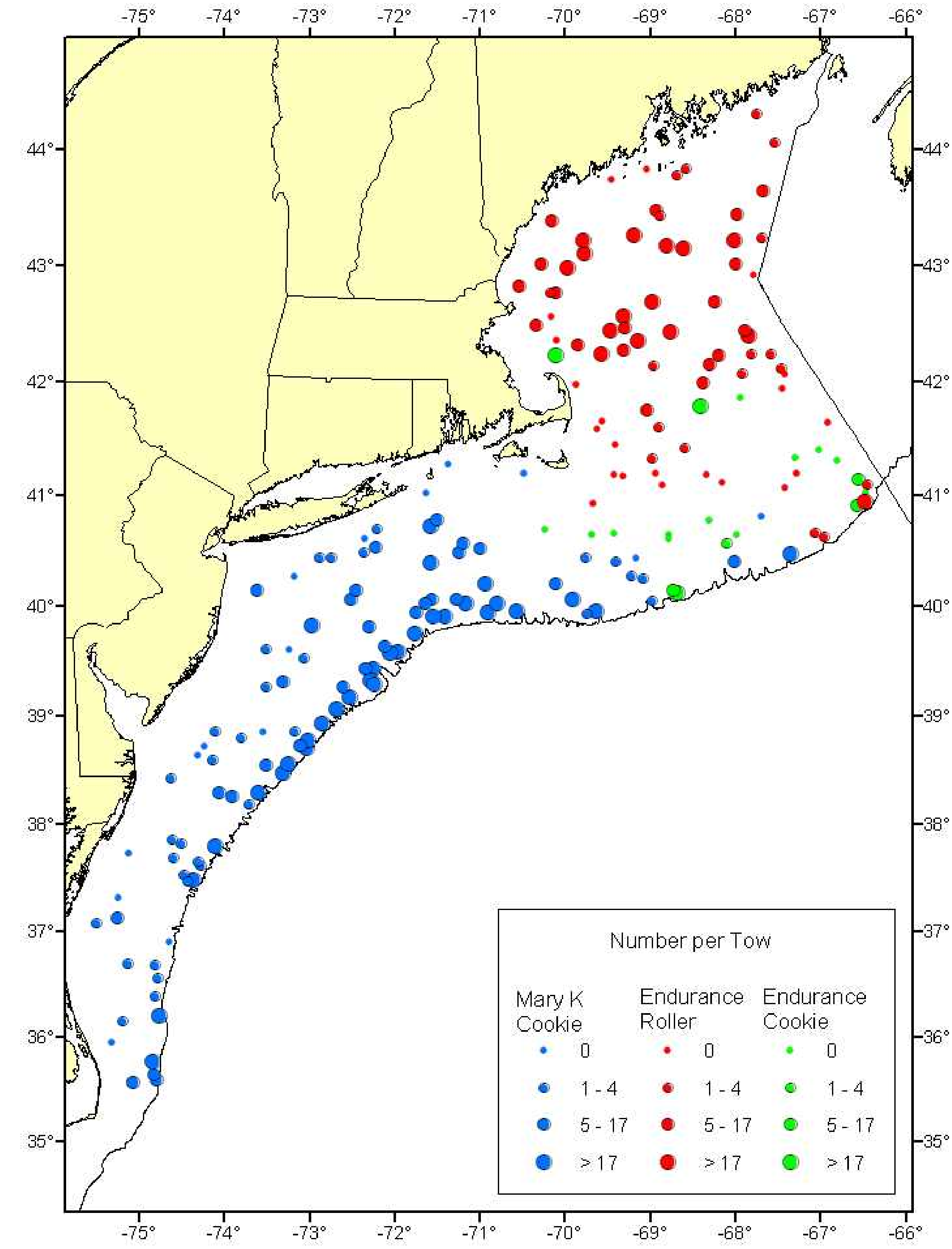
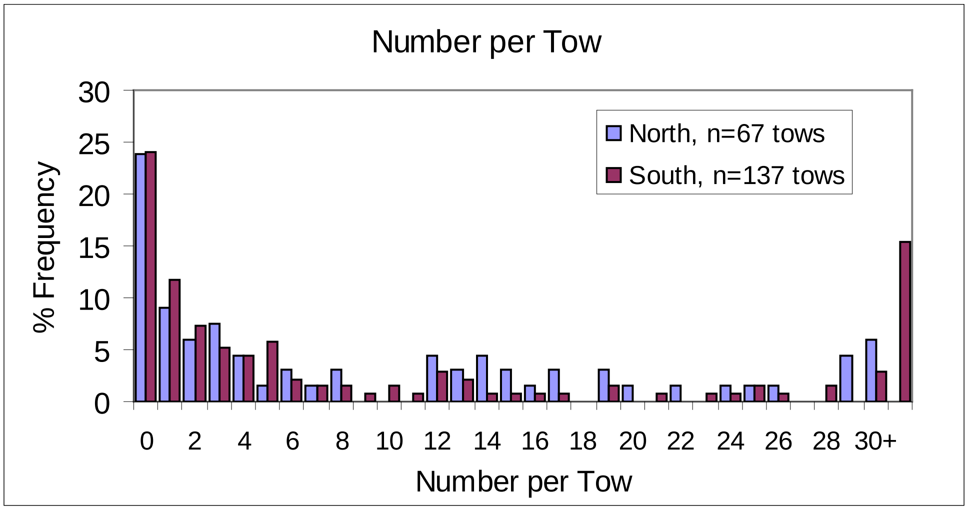


Figure A25. Nominal catch per tow (numbers) coded by net type.



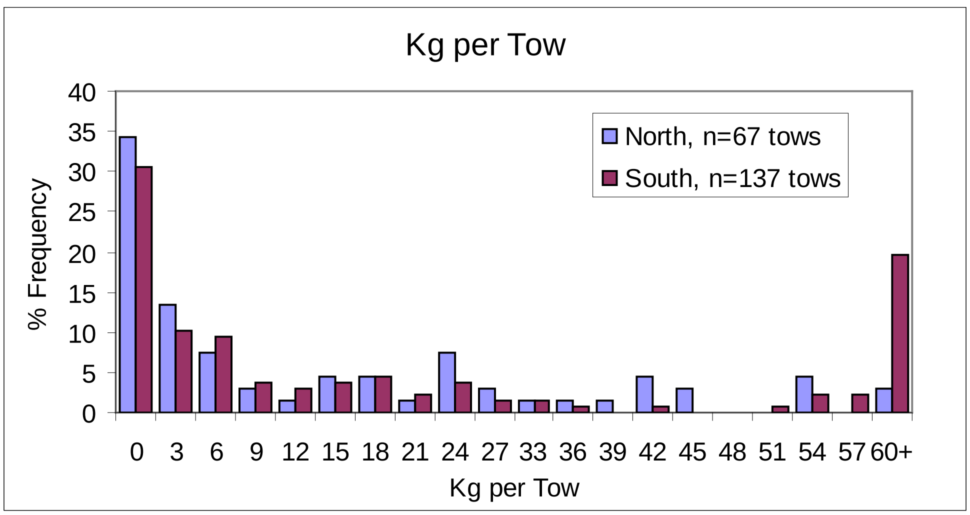
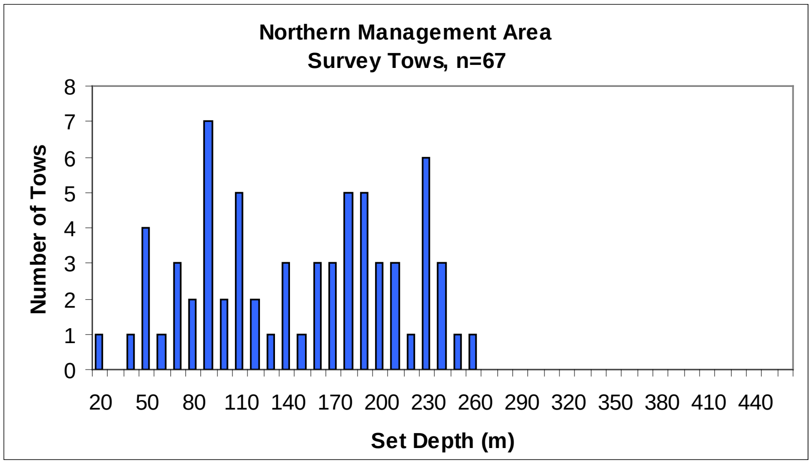


Figure A26. Relative frequency of catch per tow (number, kg), good survey tows.



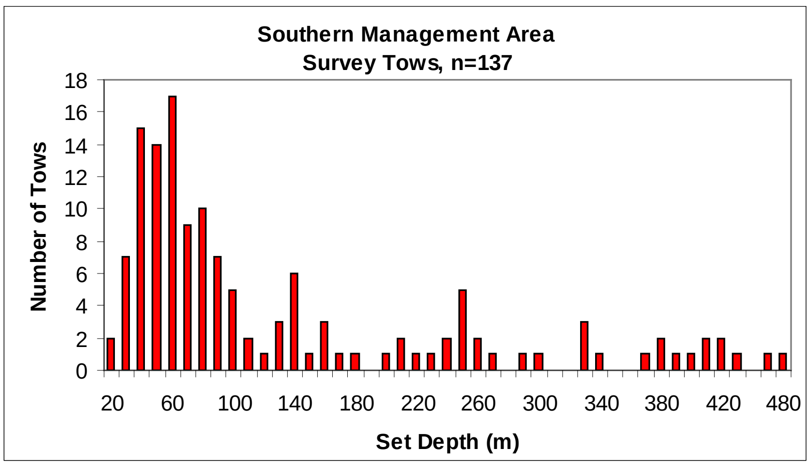


Figure A27. Depth distribution (binned by 10 m) of good survey tows from 2009 cooperative monkfish survey.

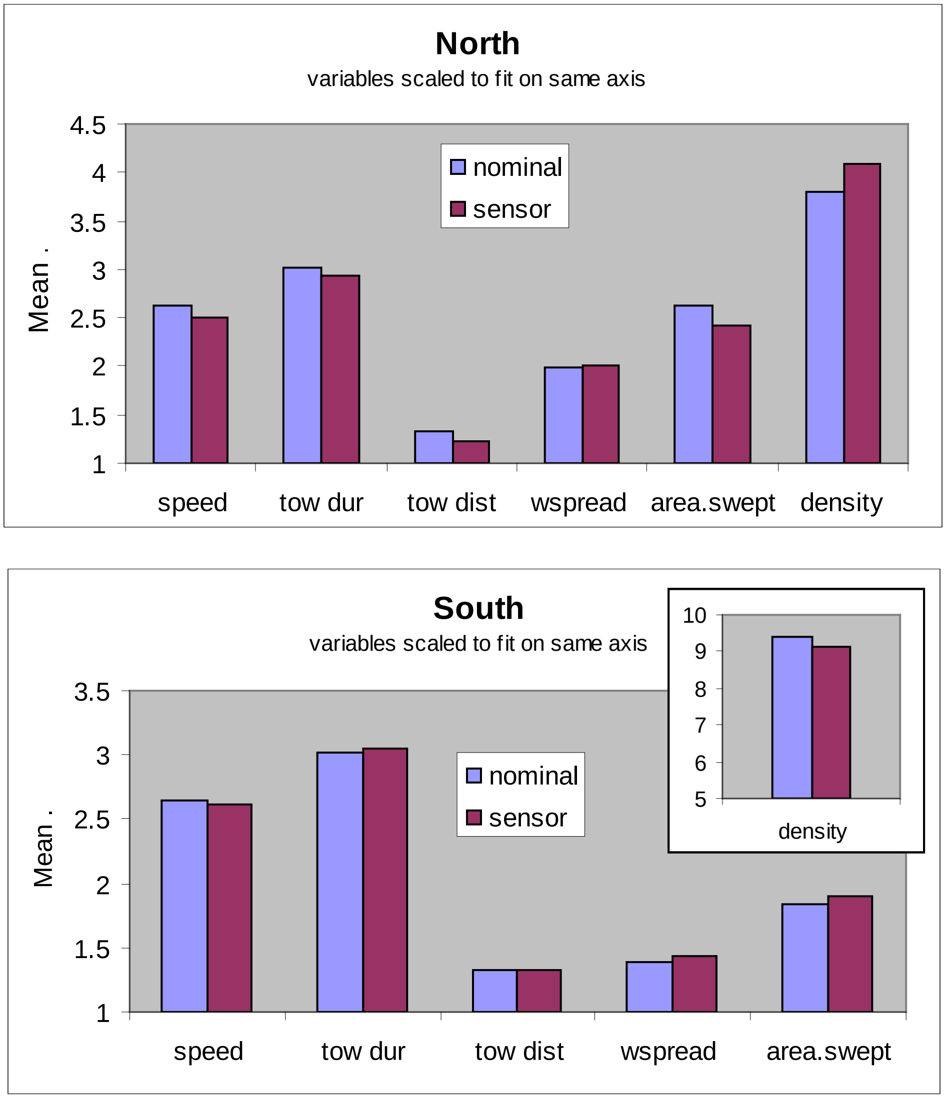
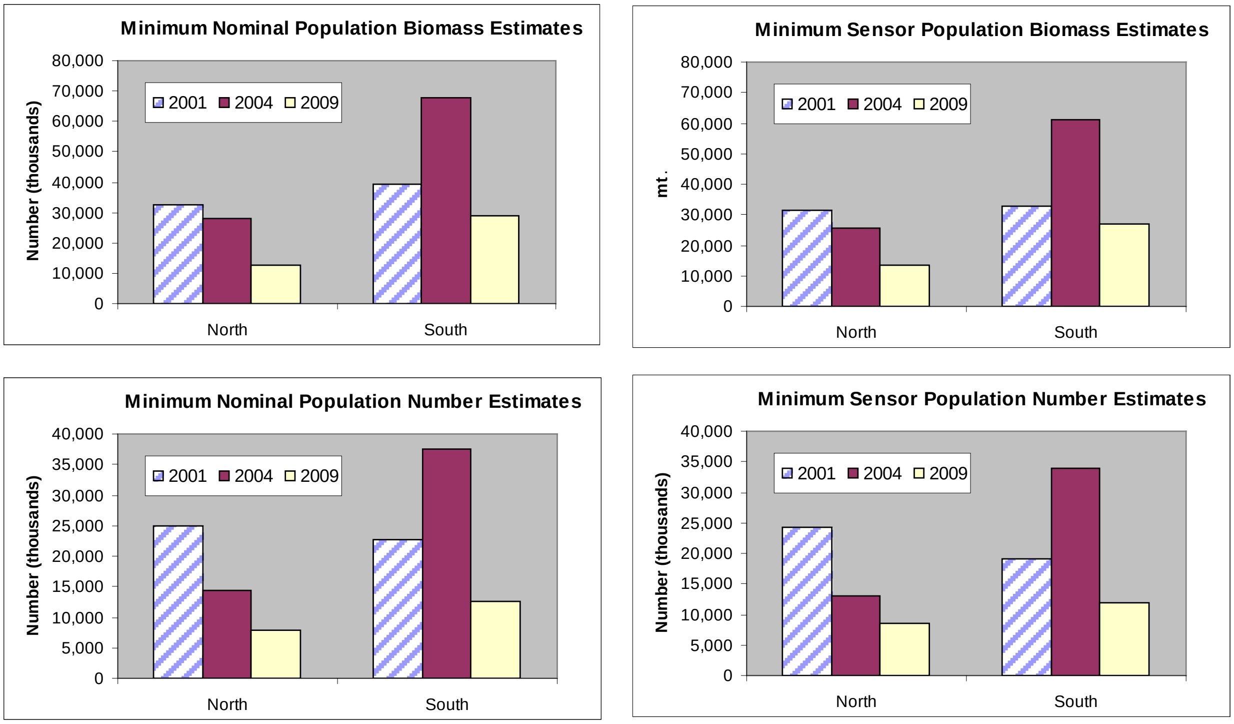


Figure A28. Effects of adjustments to derive sensor-based estimates, averaged over management area. Variables have been scaled to fit on the same x-axis. Density is number per nautical mile.

A.



B.

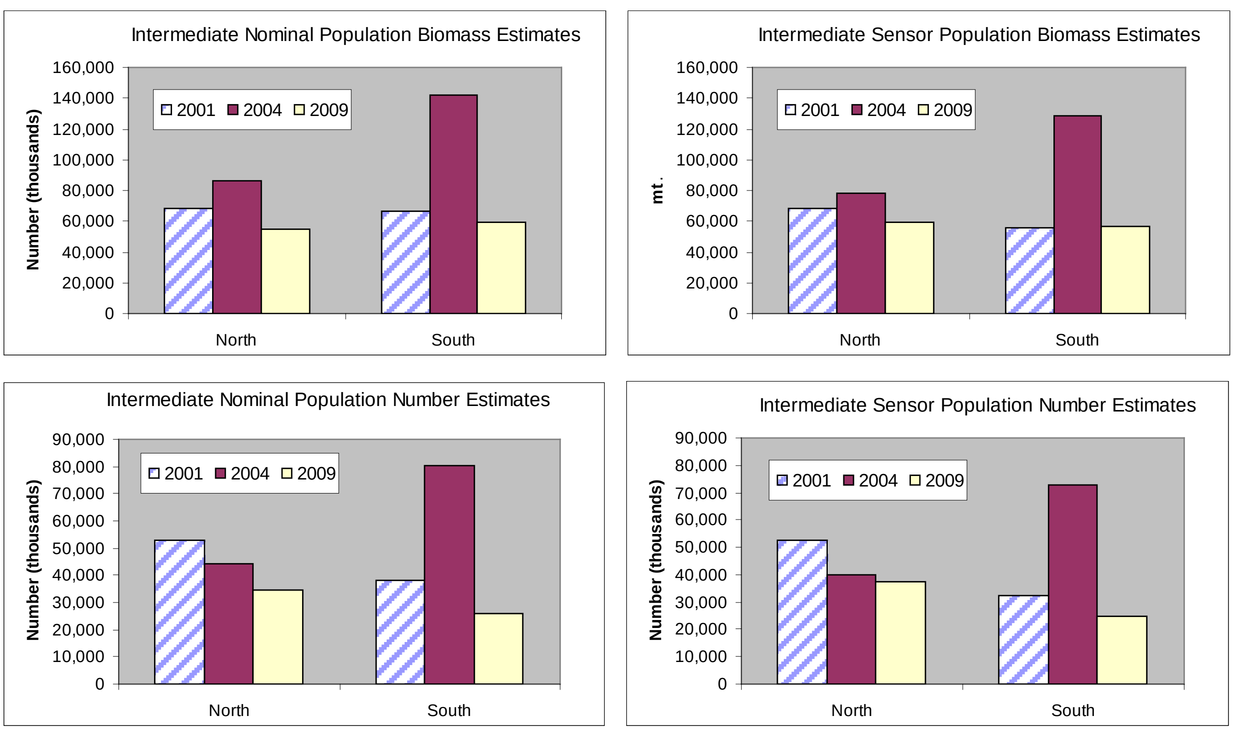


Figure A29. A. Minimum estimates (assuming 100% net efficiency) of population size and biomass based on nominal and sensor-defined tows from 2001, 2004 and 2009 cooperative surveys. B. Estimates assuming intermediate net efficiencies.

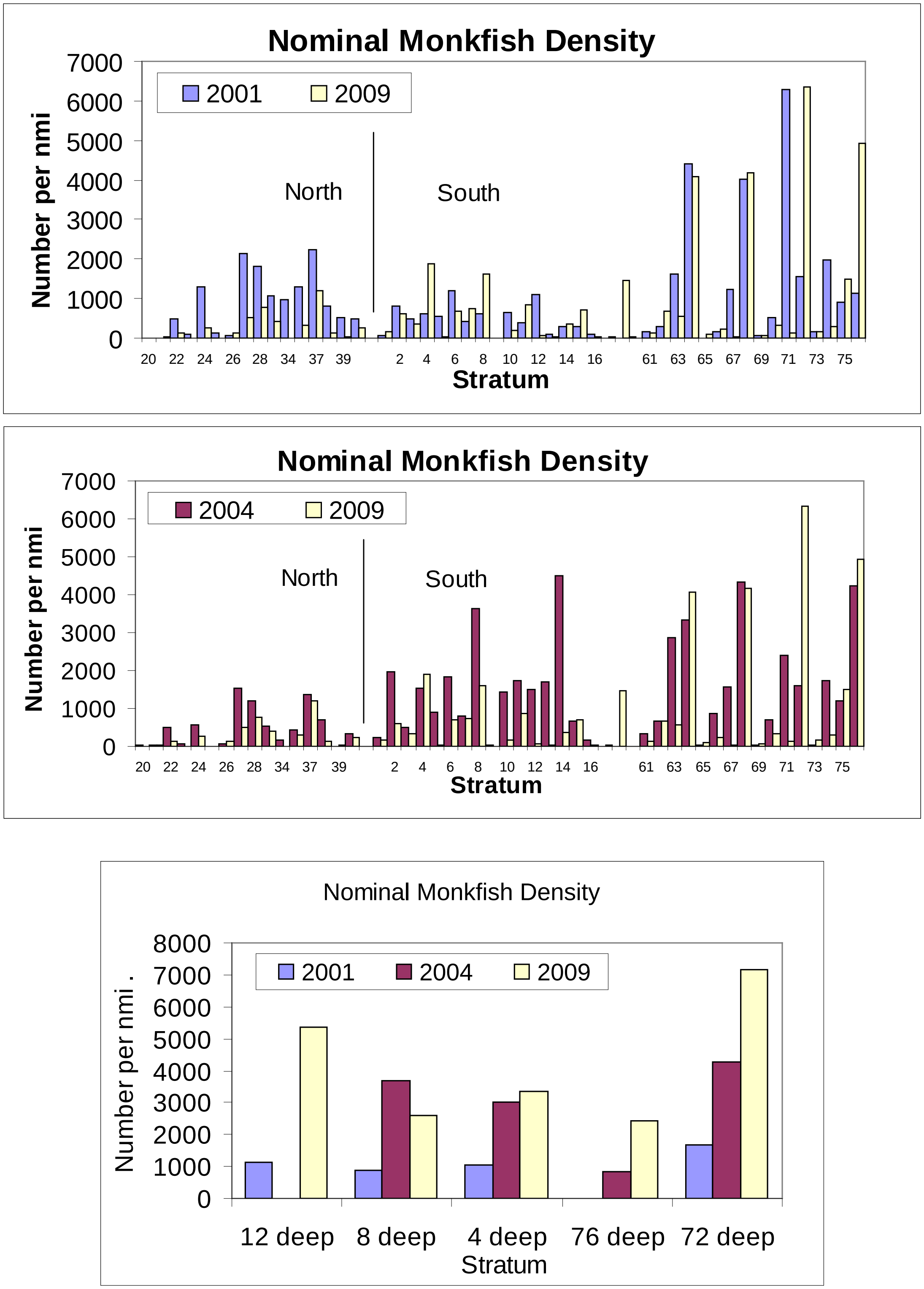


Figure A30. Monkfish density (nominal number per nmi swept) by stratum, 2001, 2004, and 2009 cooperative surveys. Bottom panel shows deep strata.

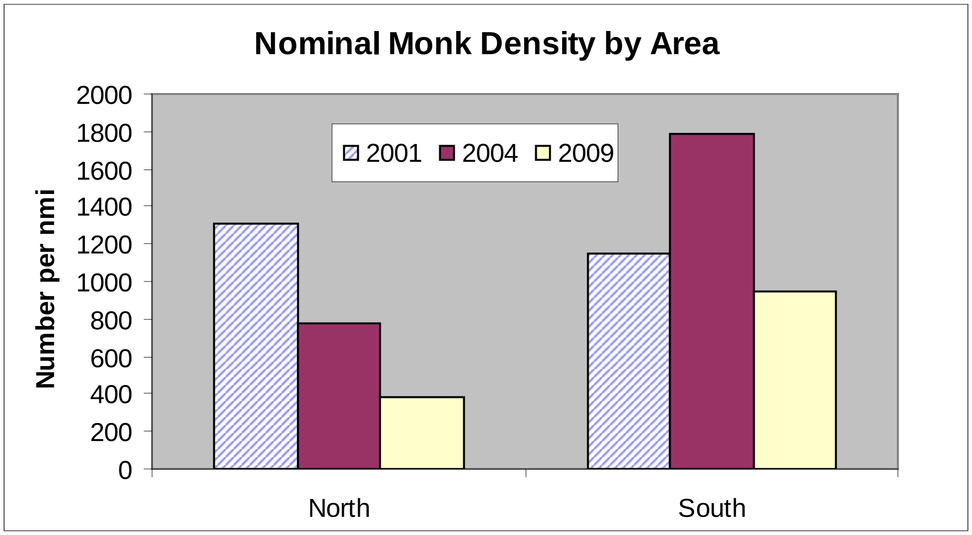
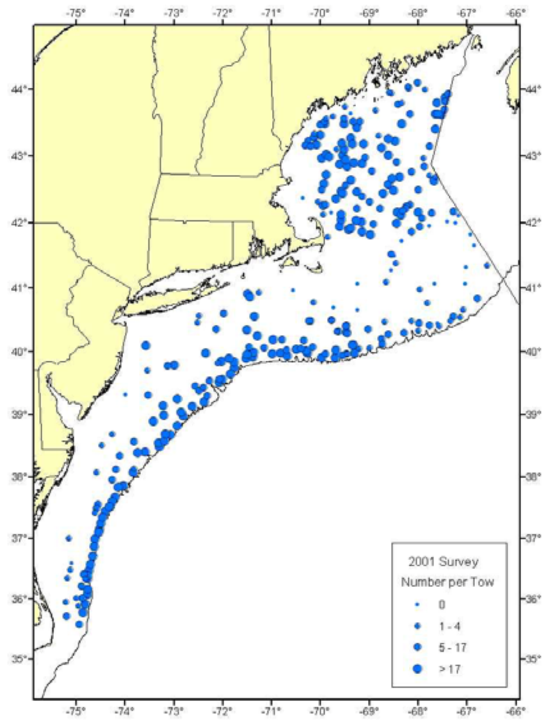
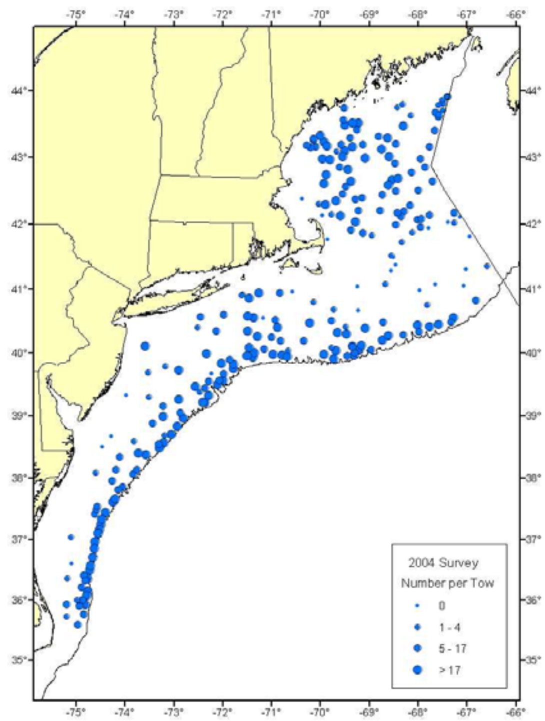


Figure A31. Nominal monkfish density by management region, 2001, 2004, 2009.


2001 Survey
Number per Tow
0
1 - 4
5 - 17
> 17


2004 Survey
Number per Tow
0
1 - 4
5 - 17
> 17

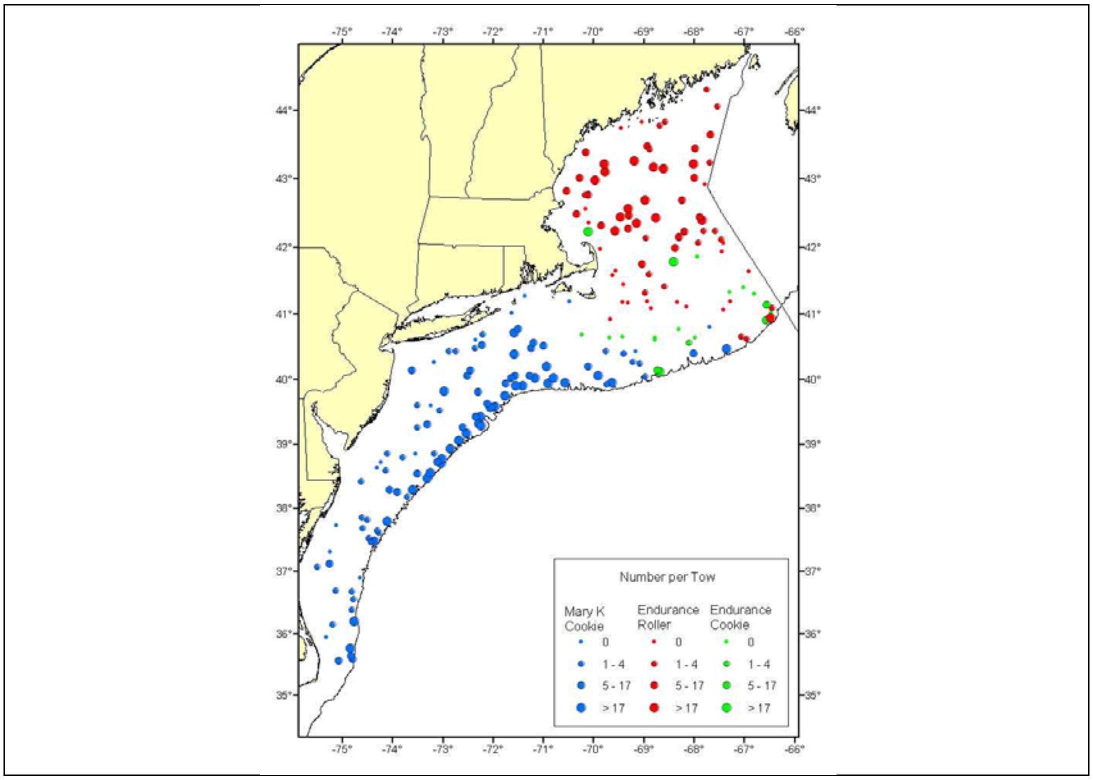


Figure A32. Distribution of catch rates (number per tow) in 2001, 2004 and 2009 cooperative surveys.

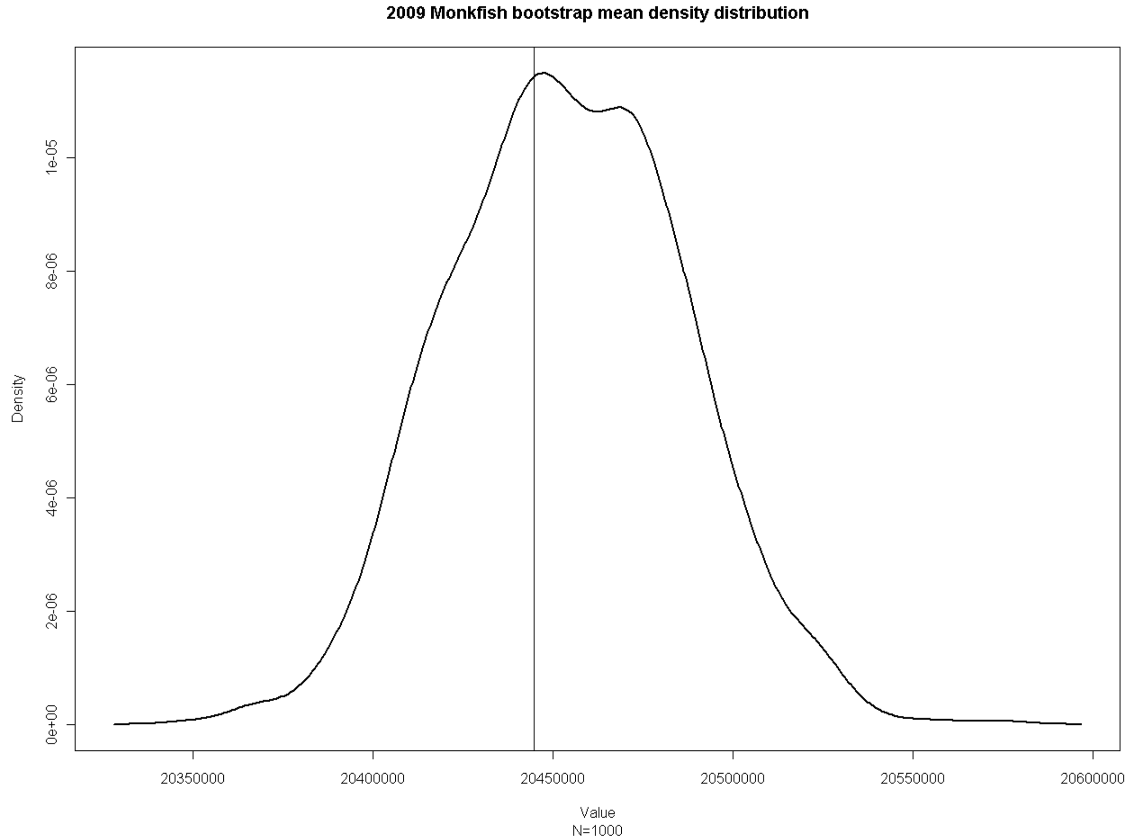
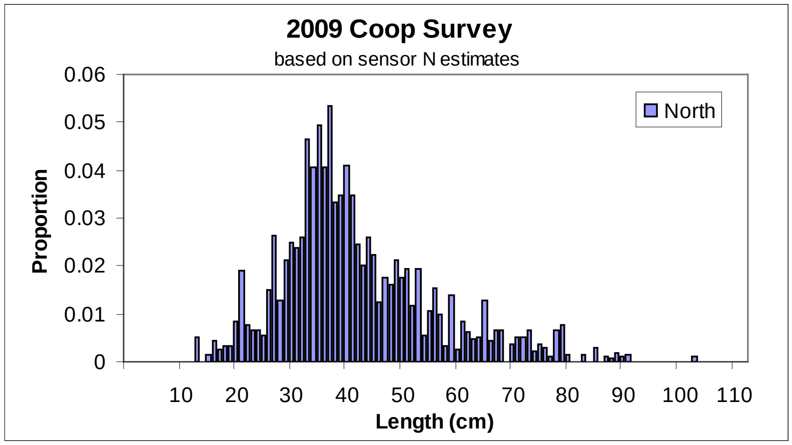
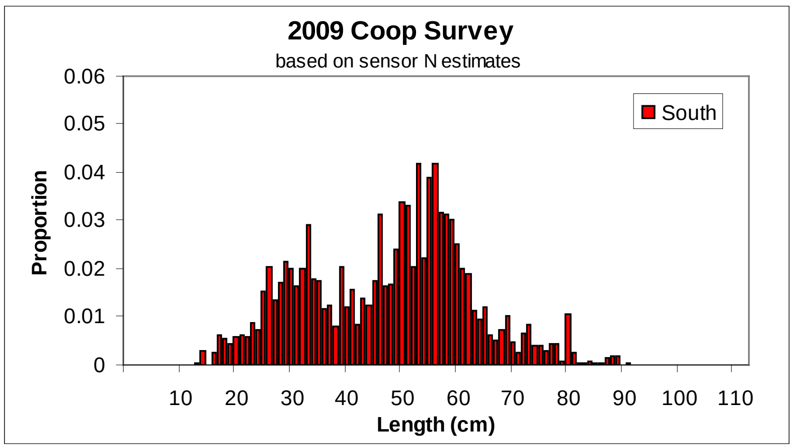
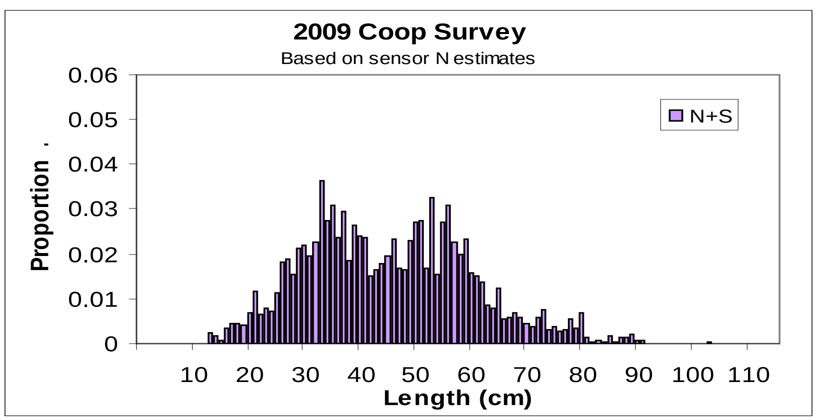


Figure A33. Results of bootstrapping analysis of monkfish catches in 2009 cooperative survey for the entire survey region based on 1,000 realizations.







Mean= 20,444,757
S.E. = 32,204.99
CV= 0.157522%

Figure A34. Proportion at length from the 2009 cooperative survey for northern, southern and combined management regions. Estimates were derived by applying proportion at length in each stratum to minimum sensor-based population numbers in that stratum, summing to numbers at length in each area and calculating the overall proportion at length.

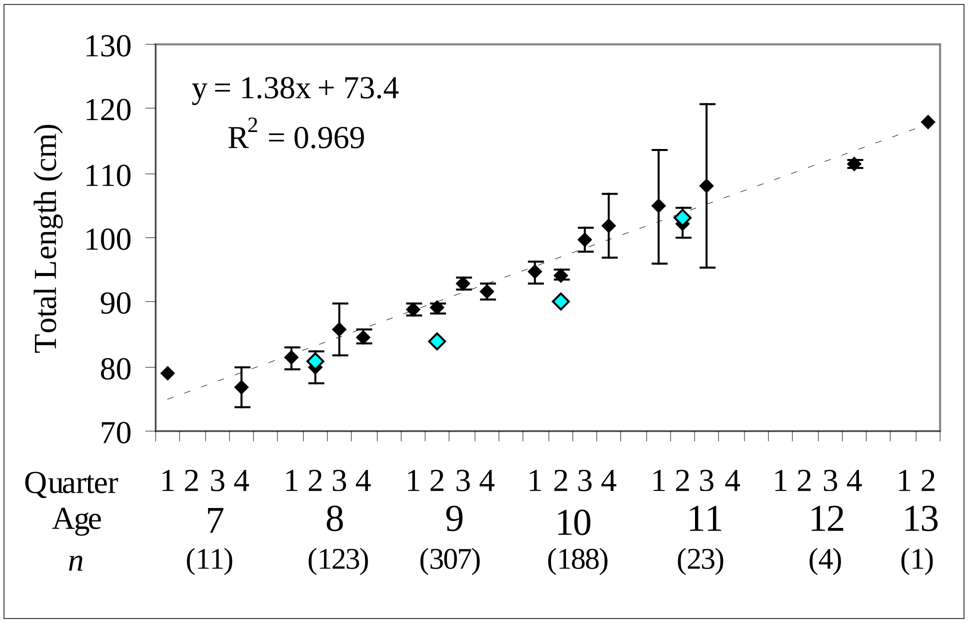


Figure A35. Mean length at age for samples from 2009 cooperative survey that were ≥ 80 cm (cyan diamonds) compared to mean length at age from NEFSC surveys and previous cooperative surveys (from Johnson et al. 2008). N for cooperative survey samples: age 8 = 4; age 9 = 17; age 10 = 3; age 11= 1.

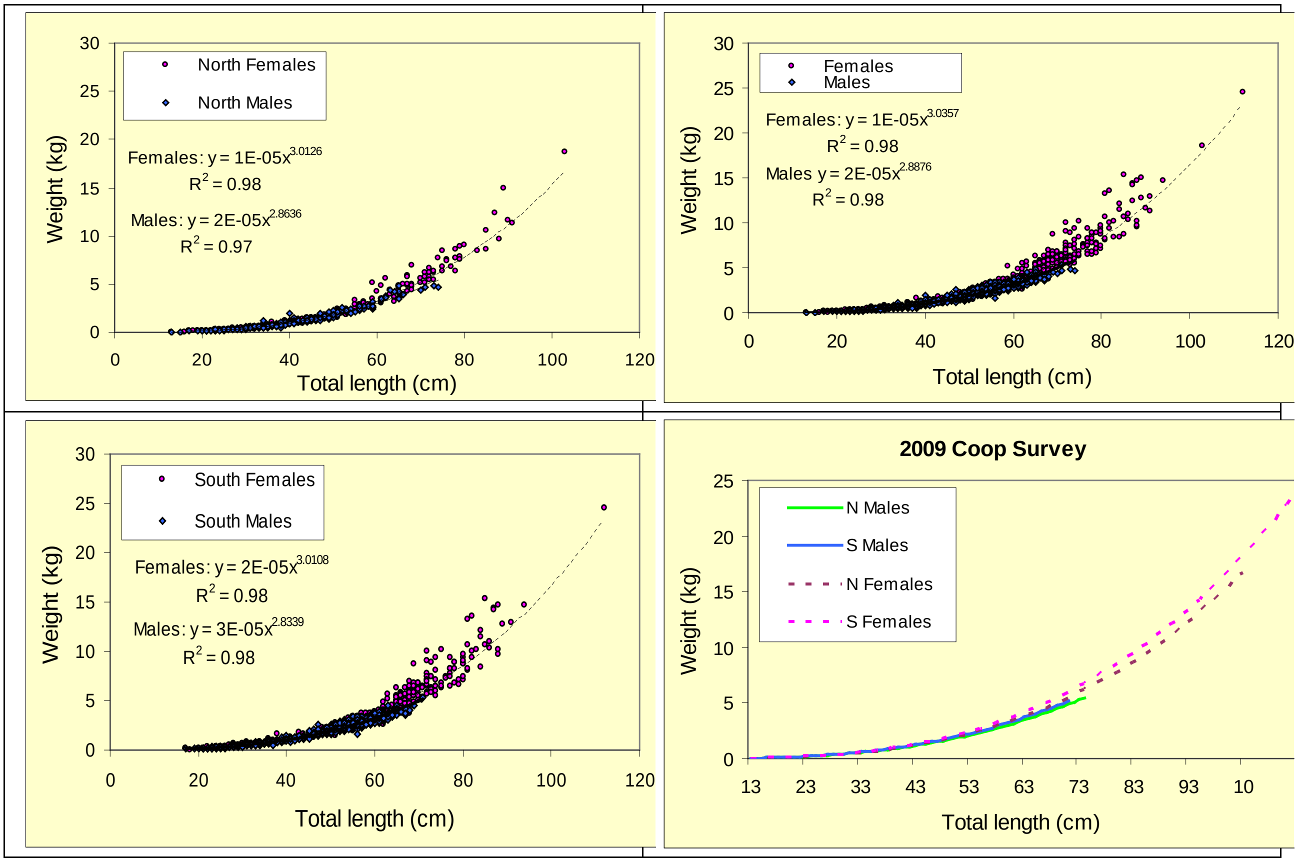


Figure A36. Length-weight relationships for male and female monkfish from northern and southern management areas, 2009 cooperative survey data.

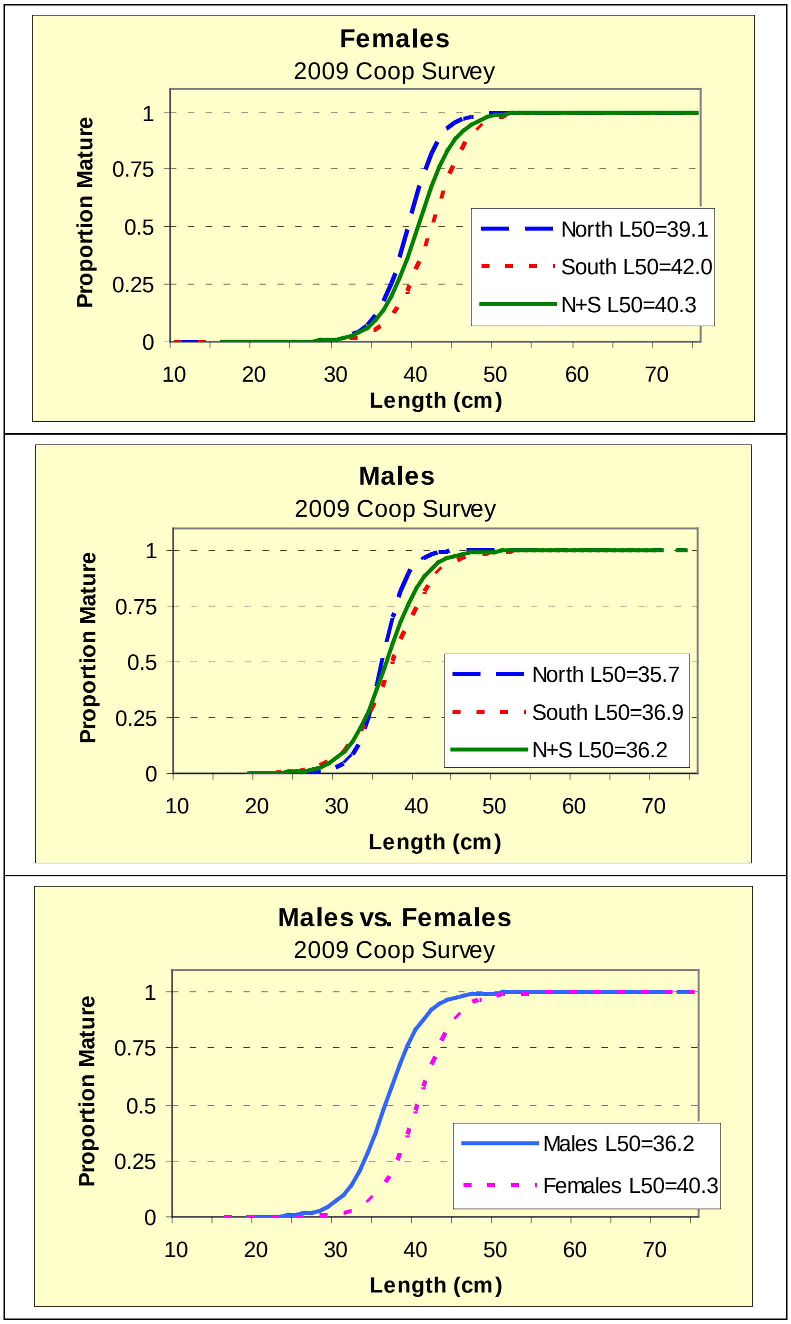


Figure A37. Maturity ogives for male and female monkfish from northern and southern management areas, 2009 cooperative survey data.

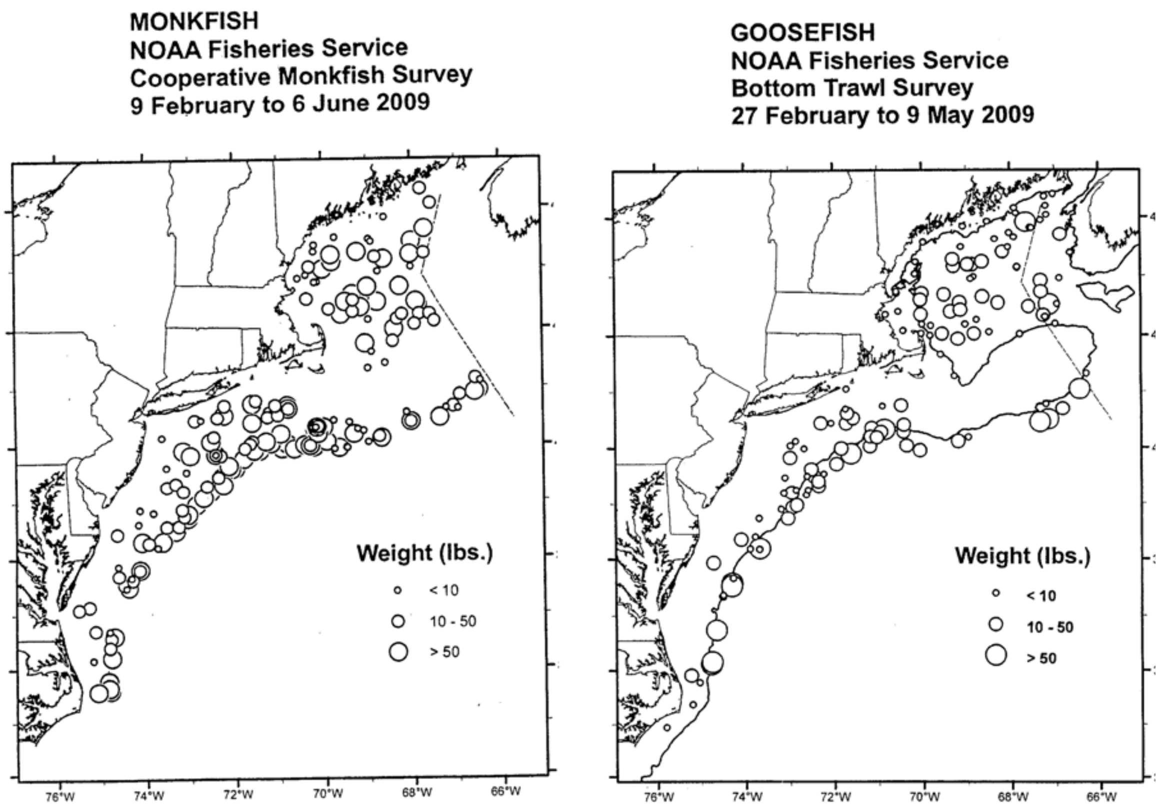


Figure A38. Catch rates (kg per tow) in 2009 cooperative monkfish survey and NEFSC 2009 spring survey.

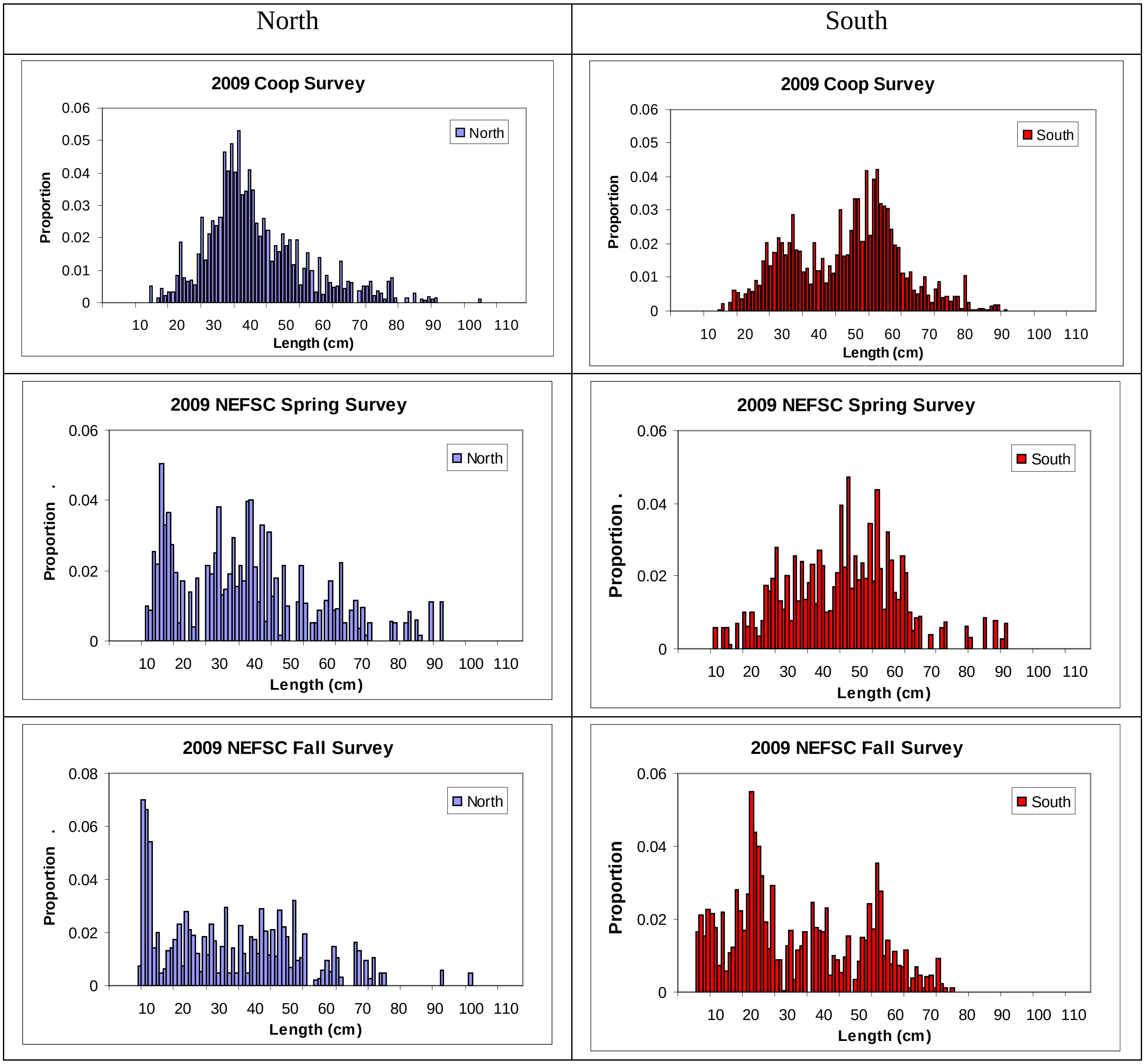


Figure A39. Proportion at length in NFMA and SFMA from the 2009 cooperative survey (top row), NEFSC 2009 spring survey (middle row) and NEFSC 2009 autumn survey (bottom row).

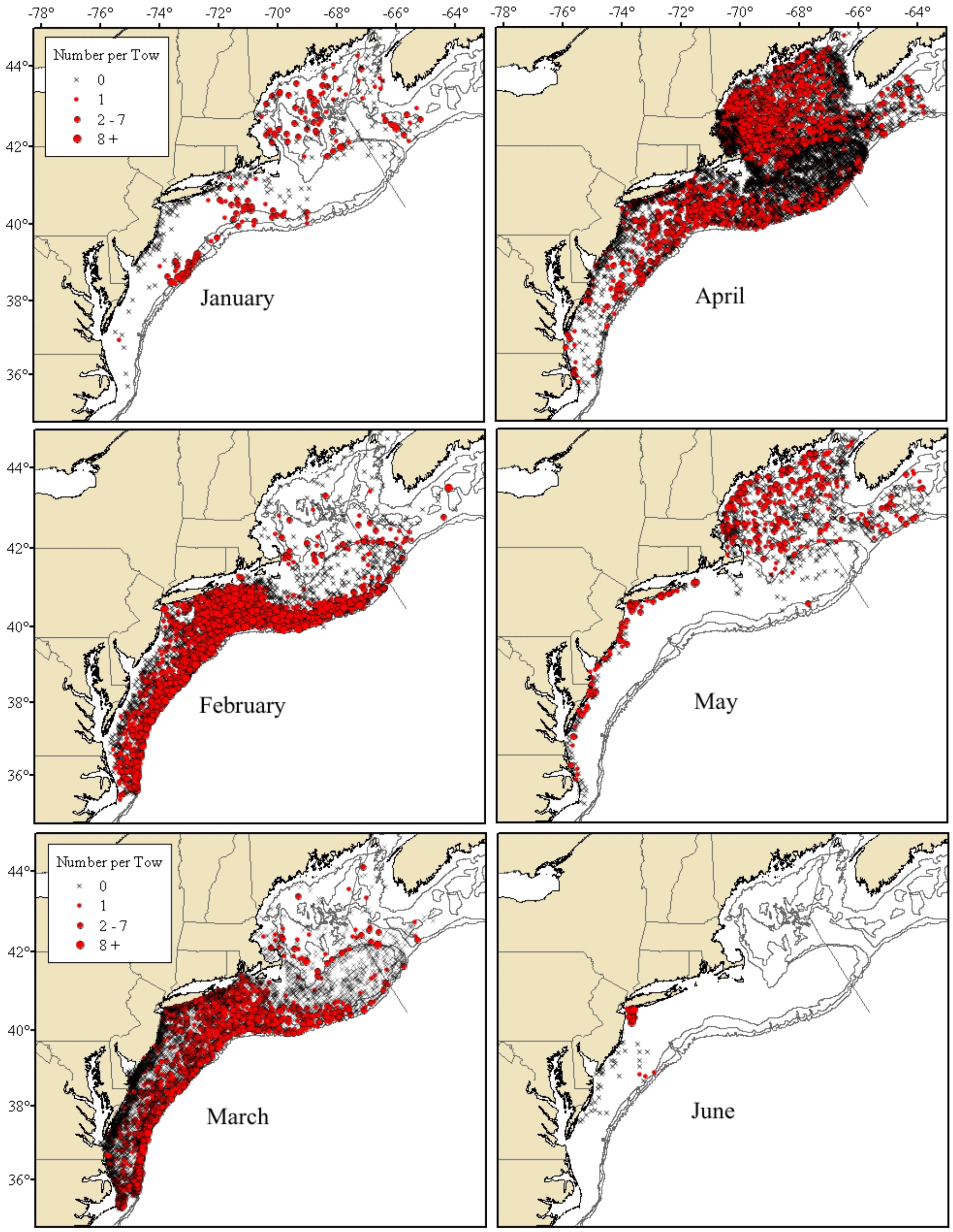


Figure A40. Monthly distribution plots for monkfish, January-December.

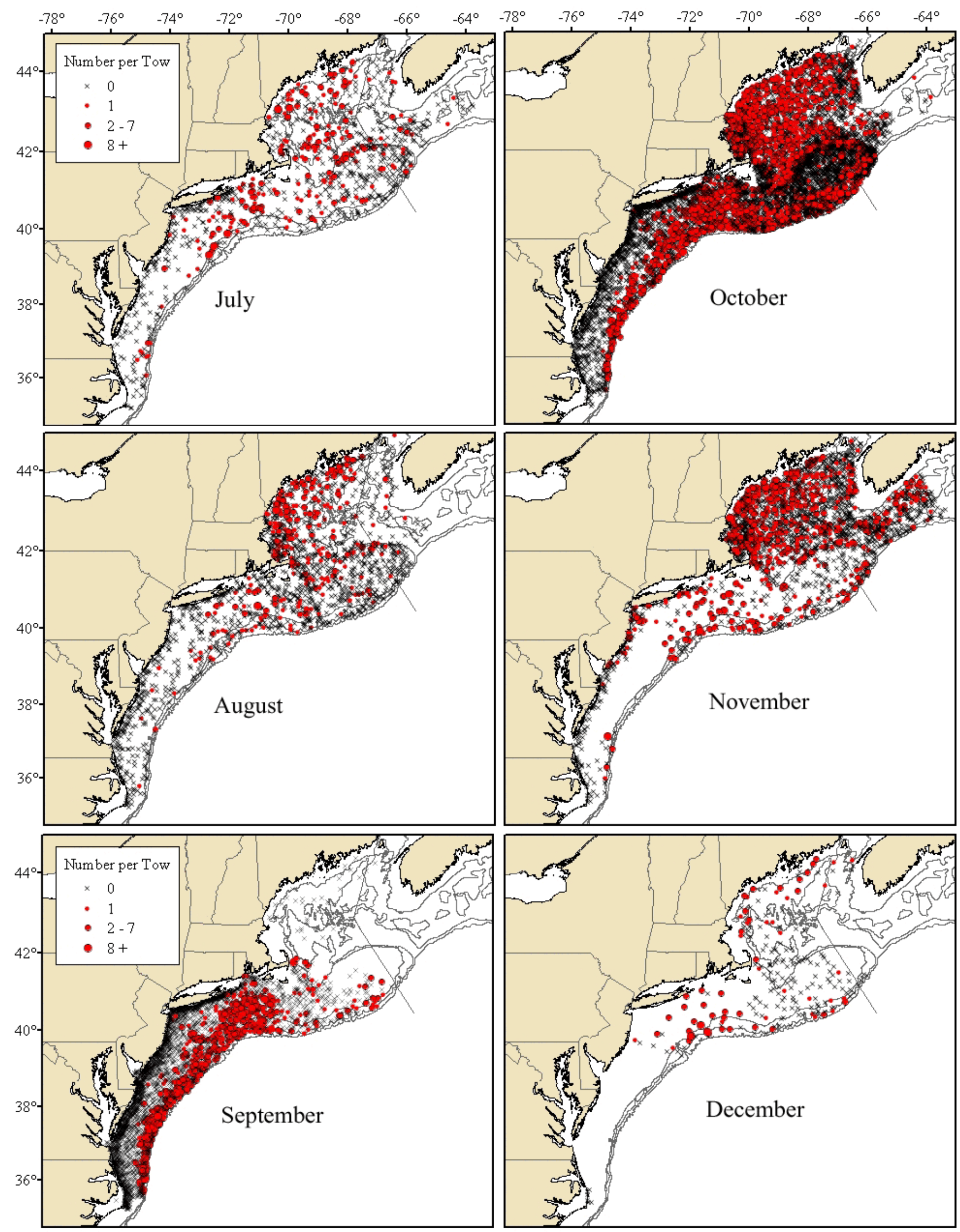


Figure A40, cont'd.

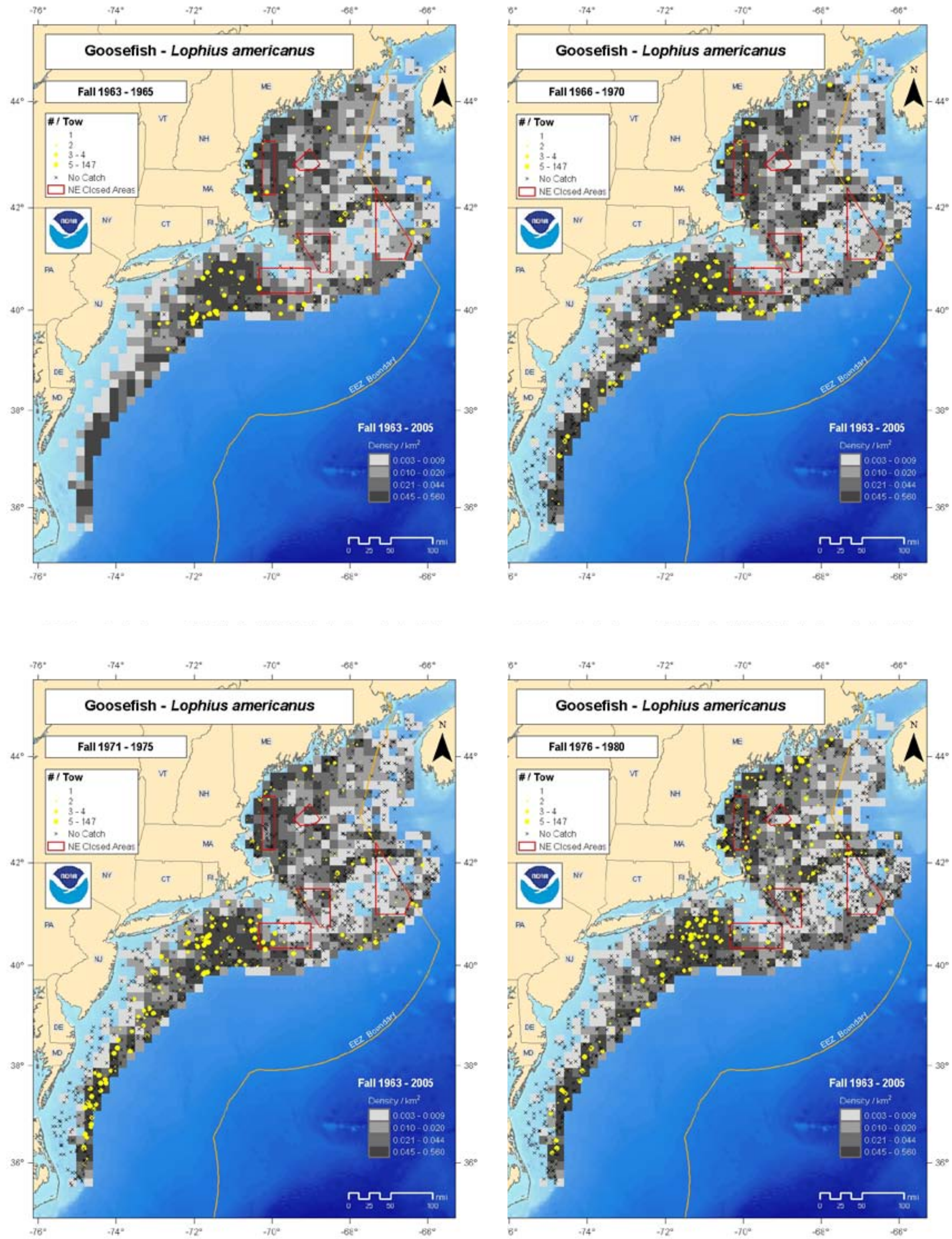


Figure A41. Distribution of monkfish from the NEFSC fall survey, 1963-2005 (from www.nefsc.noaa.gov/sos).

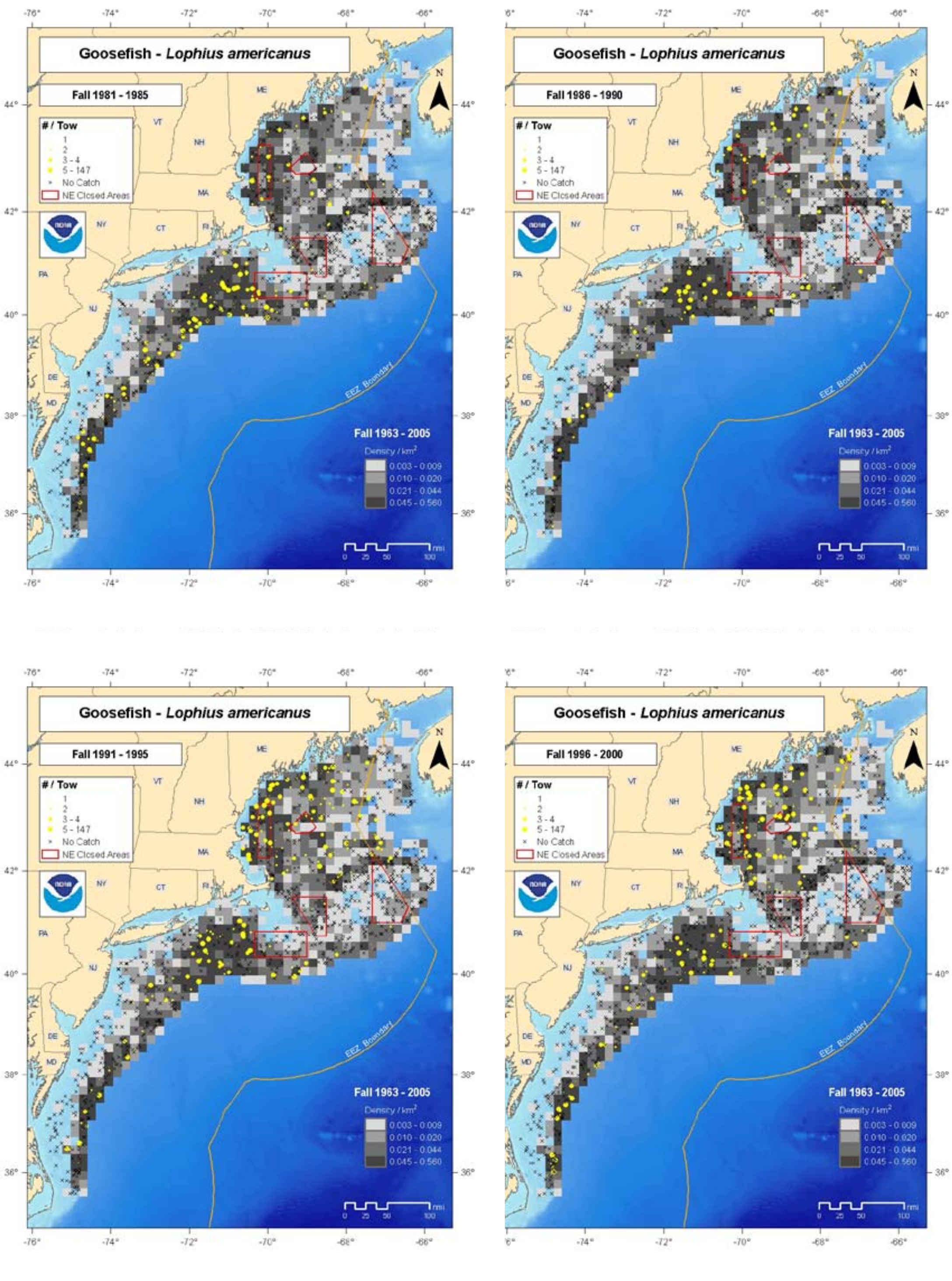


Figure A41. continued

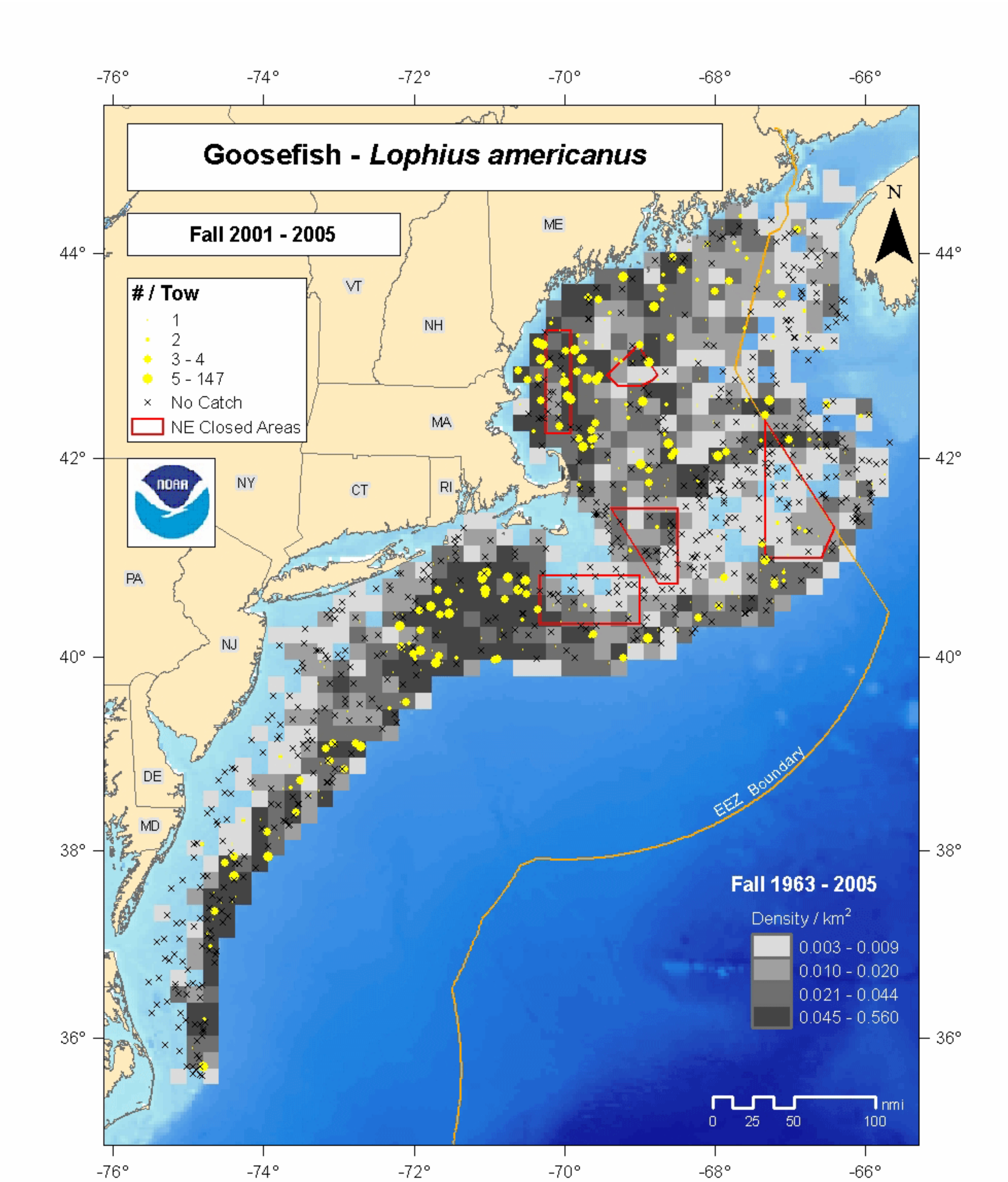
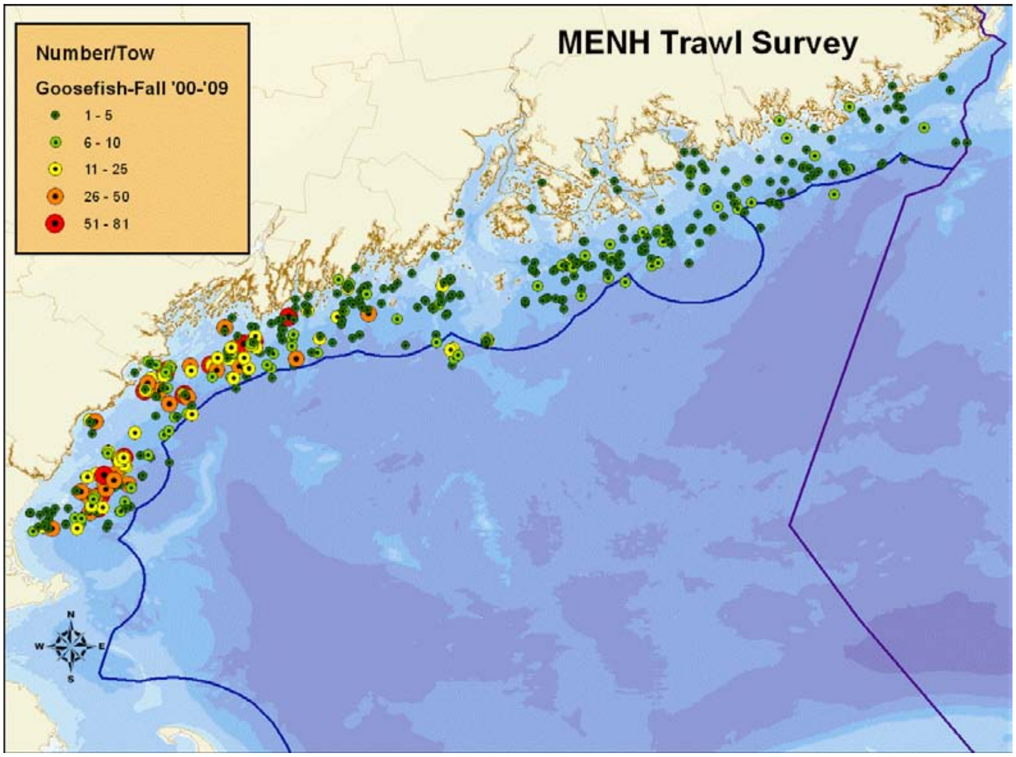


Figure A41. continued



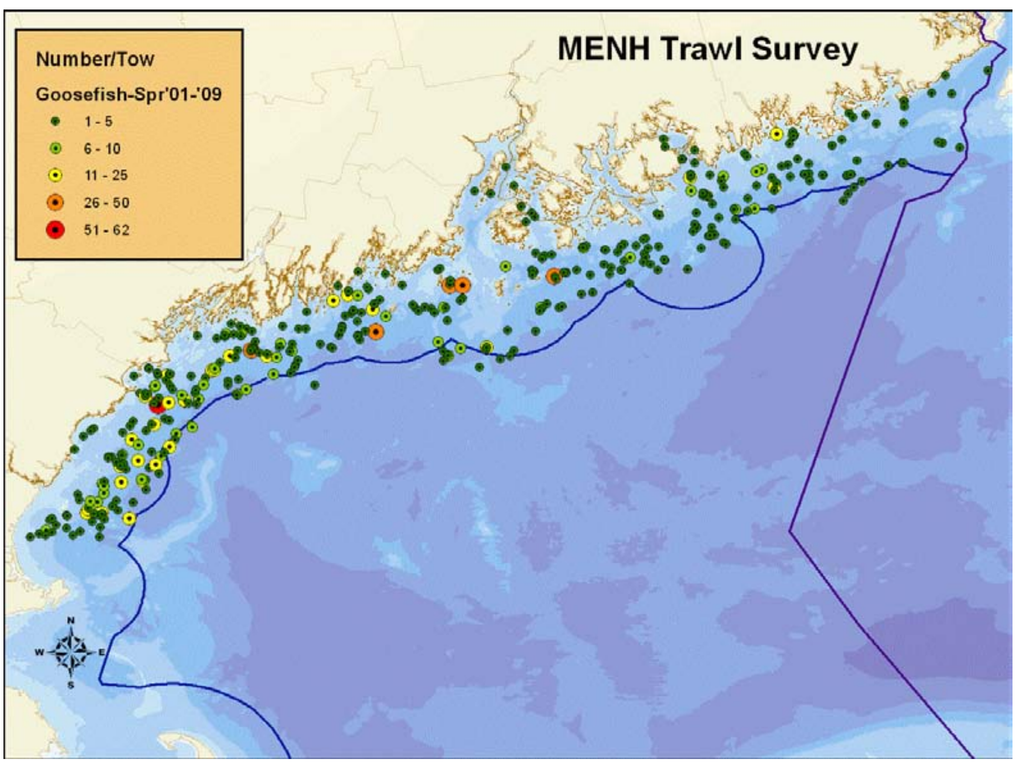


Figure A42. Geographic distribution of catches in fall (top panel) and spring (bottom panel) ME-NH inshore trawl surveys. Outer limit of survey (bold dark blue line) is 200 m.

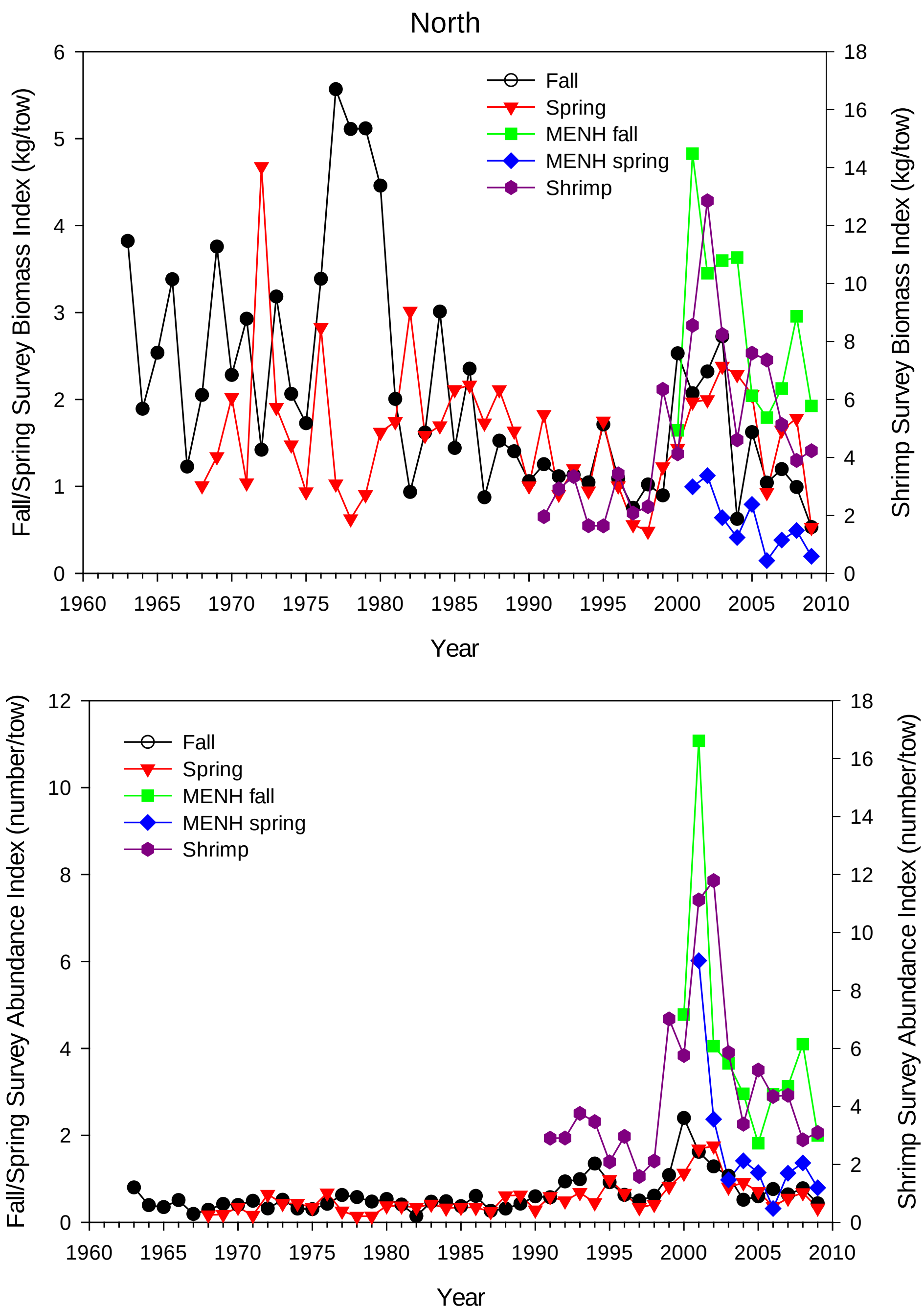


Figure A43. NEFSC spring and autumn surveys of monkfish biomass and abundance in the northern management region.

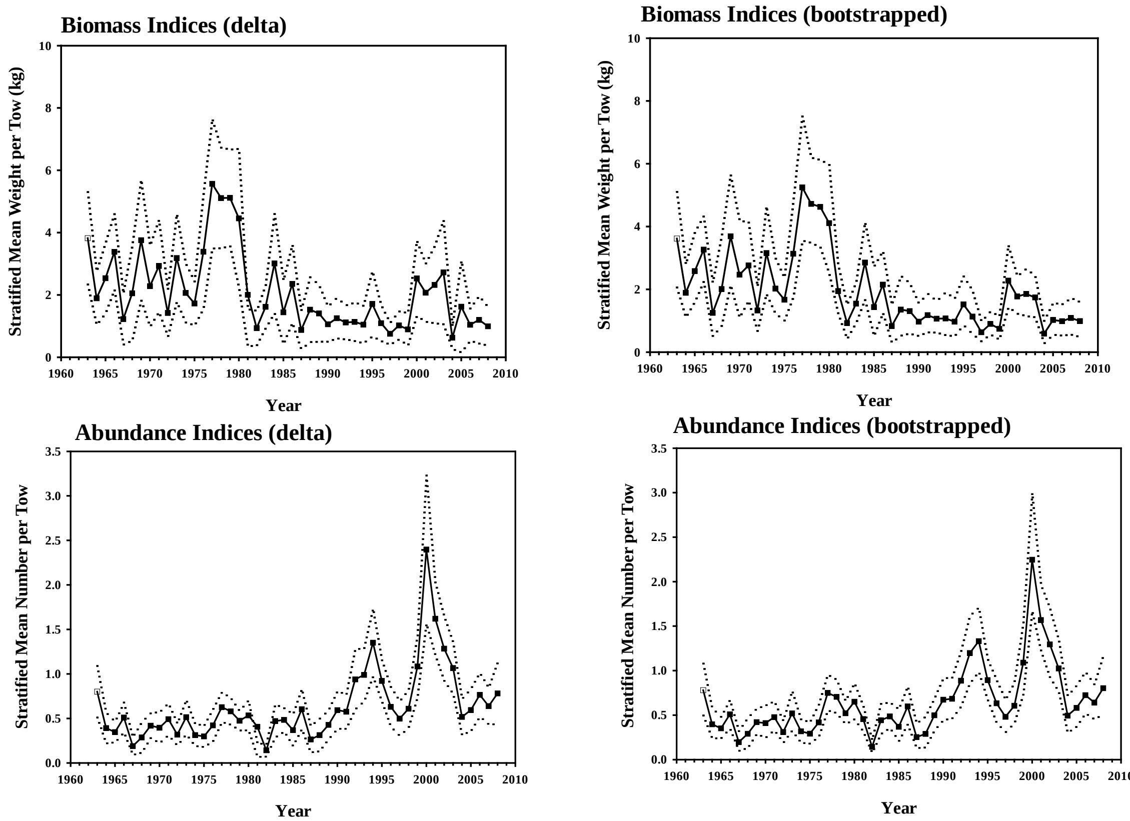


Figure A44. Delta distribution (parametric) and bootstrapped (arithmetic) biomass and abundance indices from the NEFSC autumn bottom trawl survey for the northern management region from 1963-2008. Data prior to 1971 have been revised following an audit of historical data. The 95% confidence limits are shown by the dashed line.

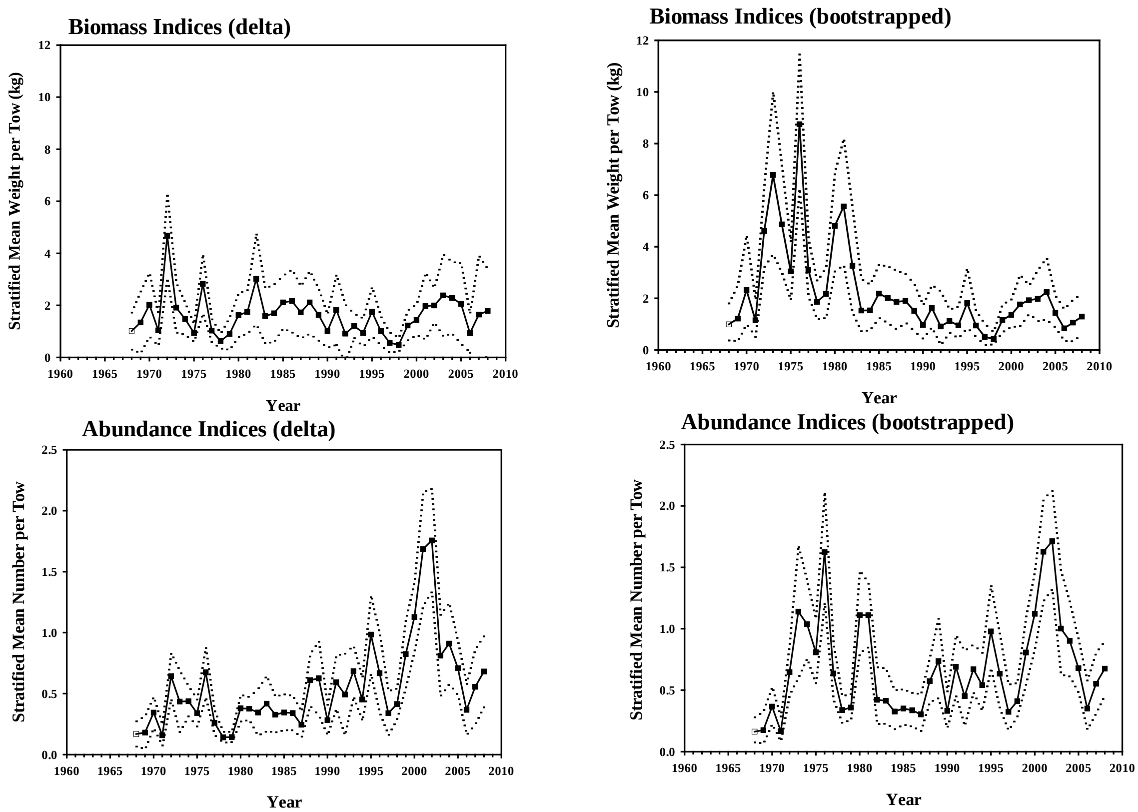


Figure A45. Delta distribution (parametric) and bootstrapped (arithmetic) biomass and abundance indices from the NEFSC spring bottom trawl survey for the northern management region from 1968-2008. Data prior to 1971 have been revised following an audit of historical data. The 95% confidence limits are shown by the dashed line.

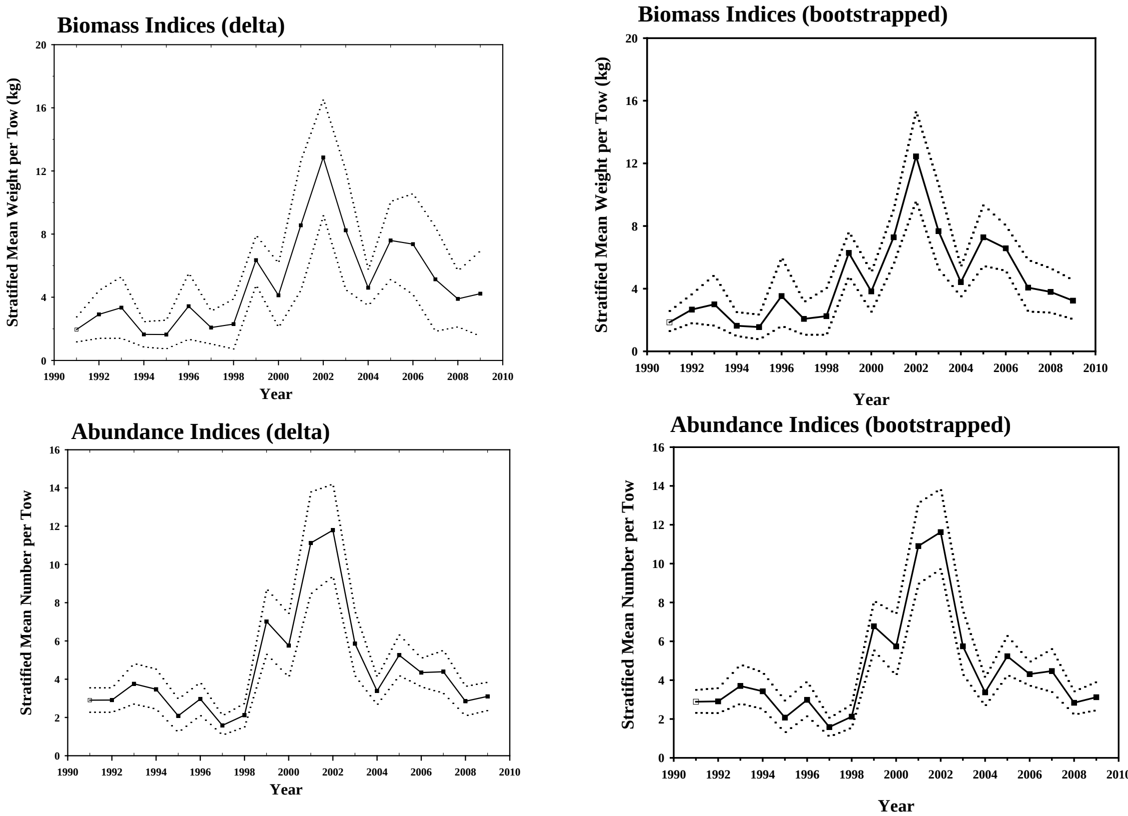
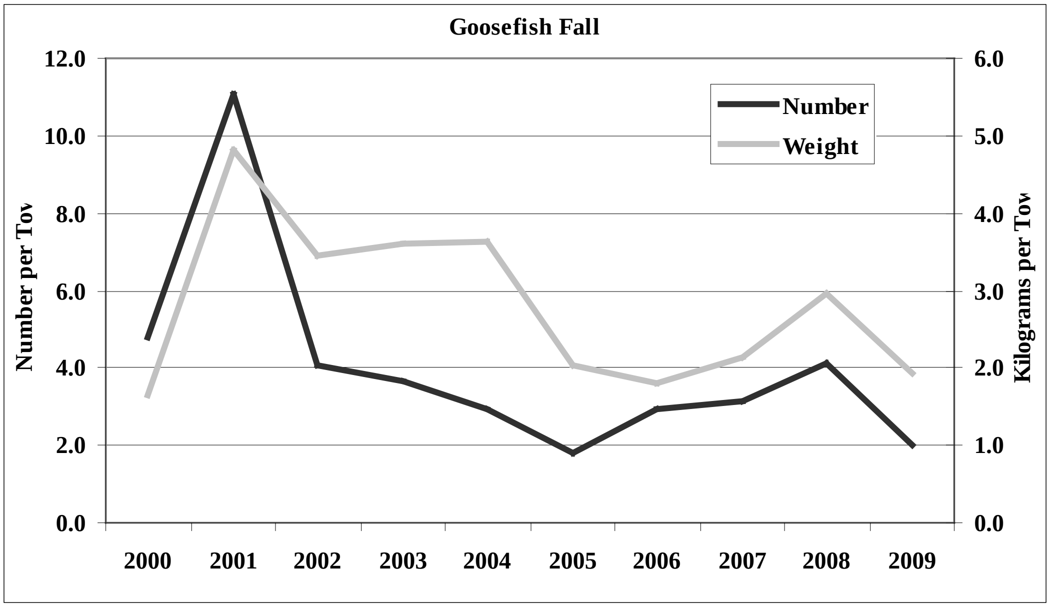


Figure A46. Delta distribution (parametric) and bootstrapped (arithmetic) biomass and abundance indices from the NEFSC shrimp survey for the northern management region from 1991-2009. Data prior to 1971 have been revised following an audit of historical data. The 95% confidence limits are shown by the dashed line.



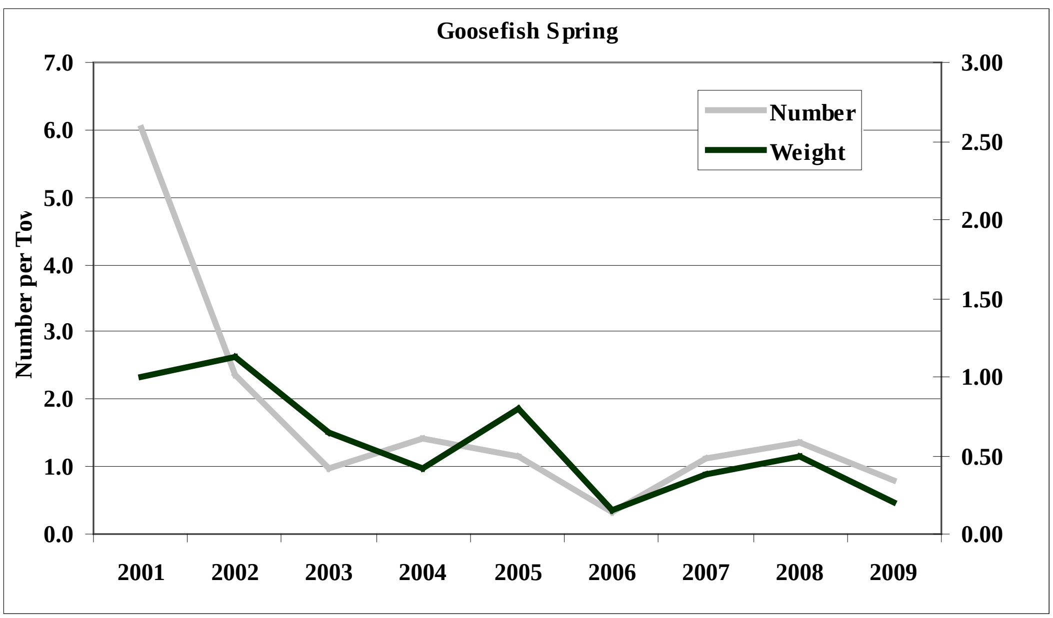


Figure A47. Survey indices from ME-NH inshore trawl surveys, NFMA.

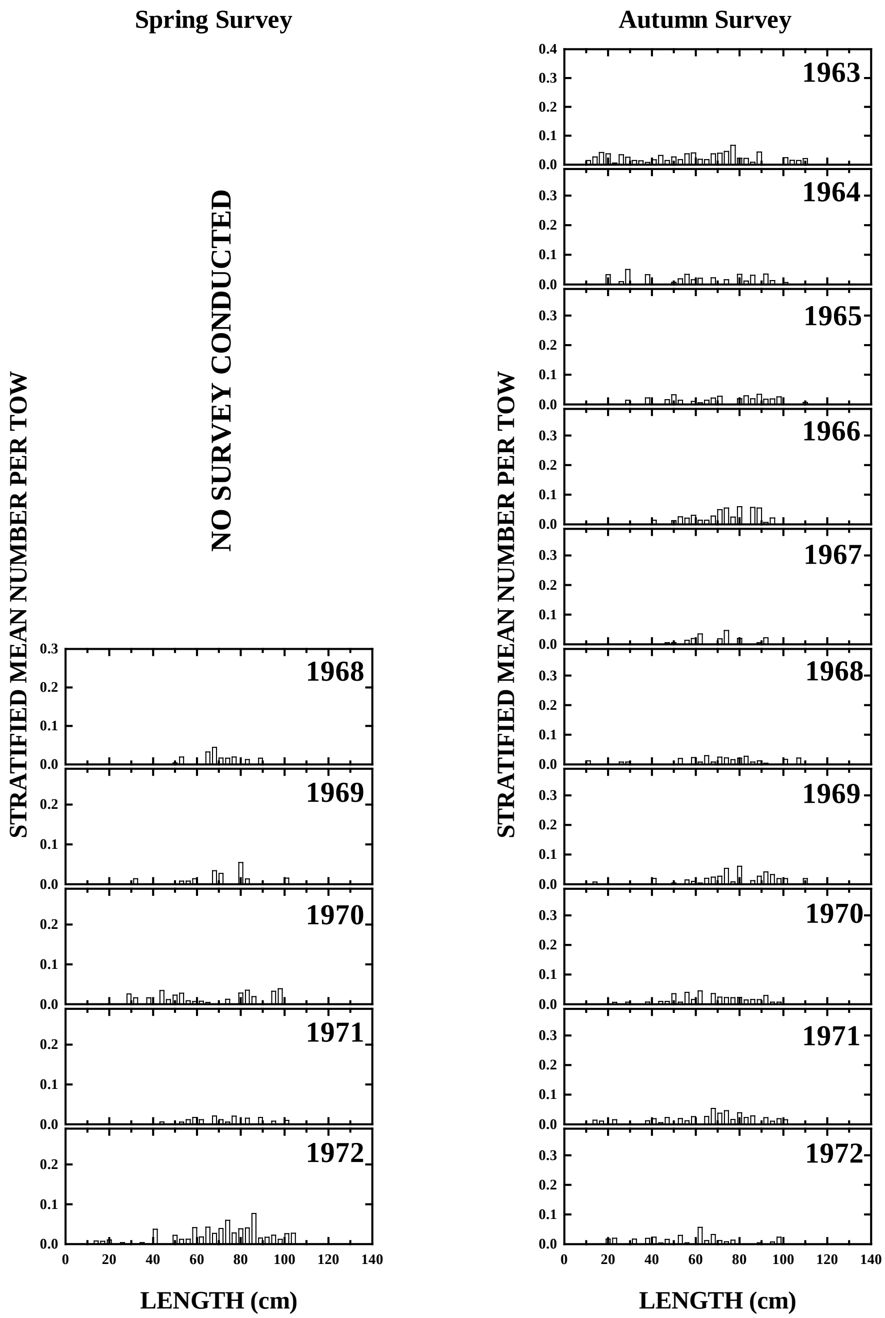


Figure A48. Goosefish length composition from the NEFSC spring and autumn bottom trawl surveys in the northern management region,

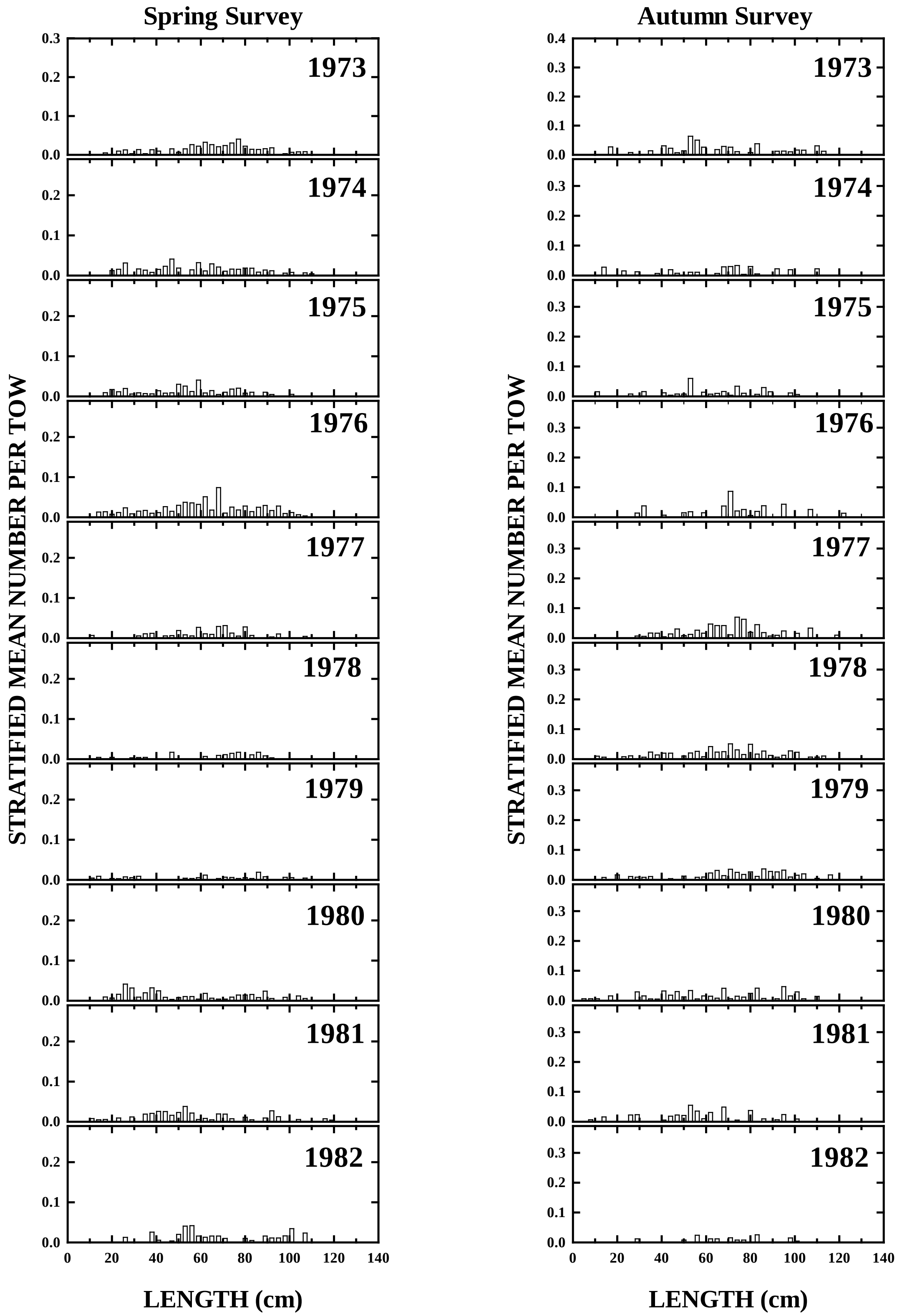


Figure A48, continued

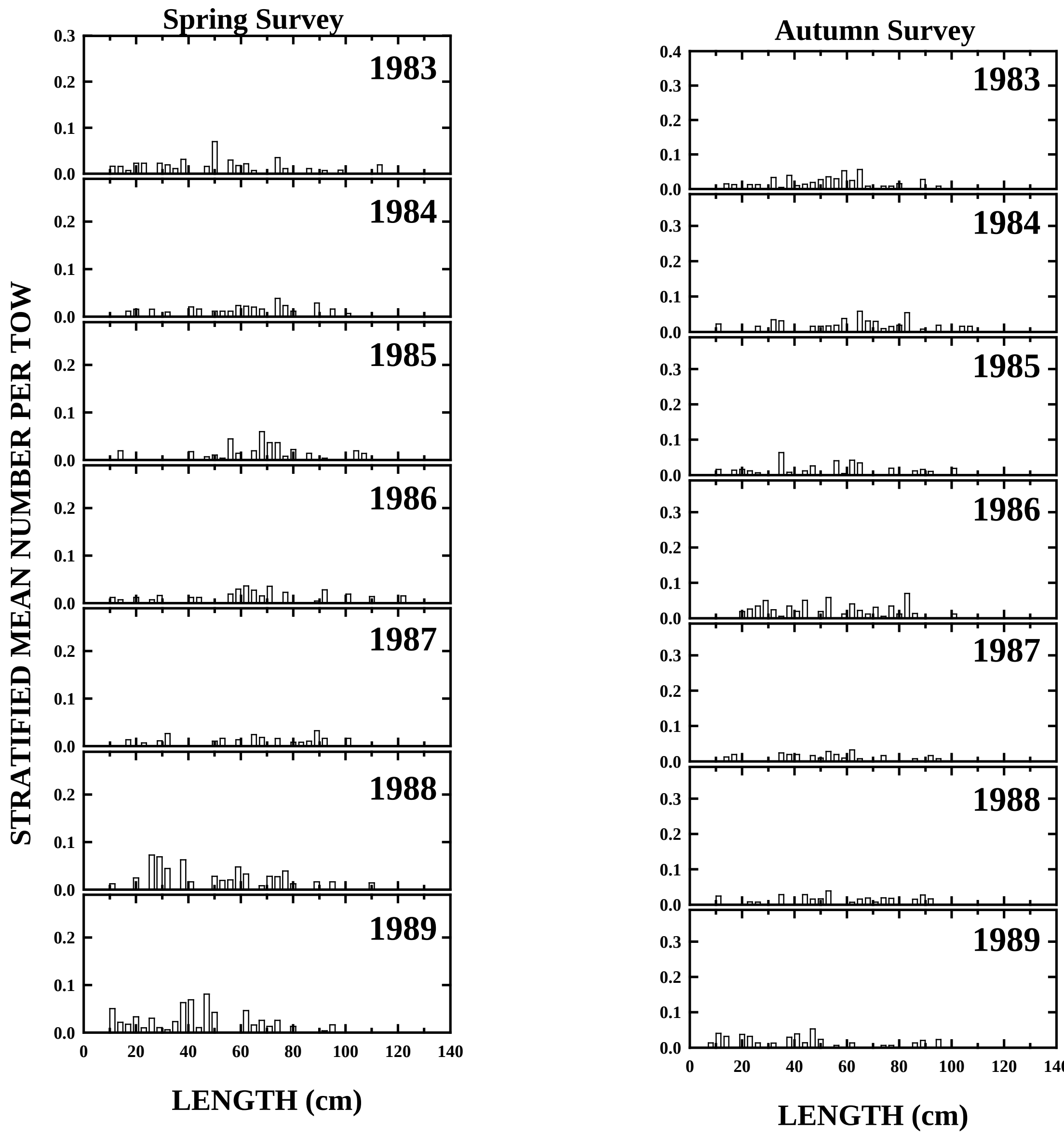


Figure A48, continued.

NOTE: Y-AXIS SCALE CHANGES ON THIS PAGE

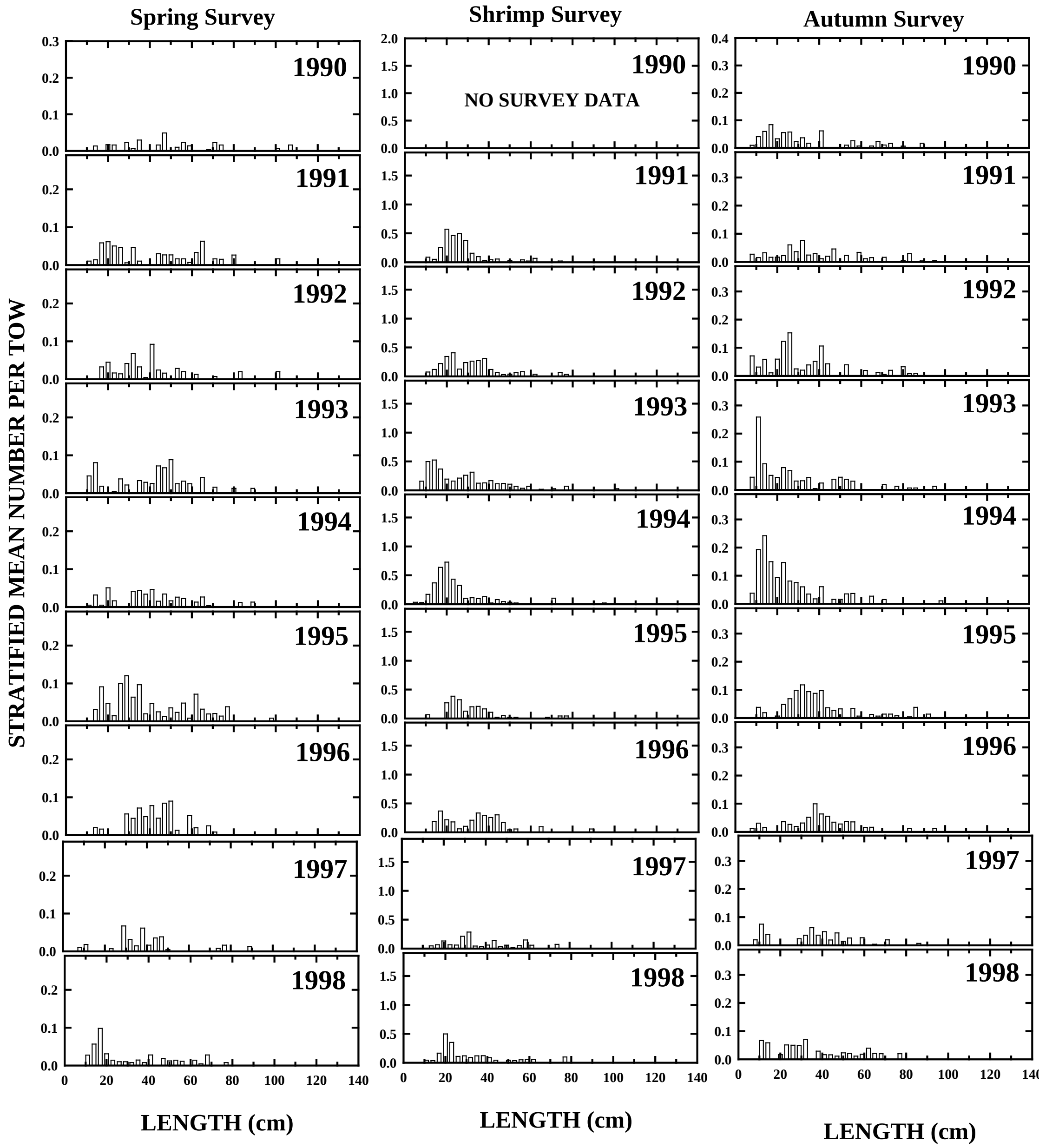


Figure A48, continued

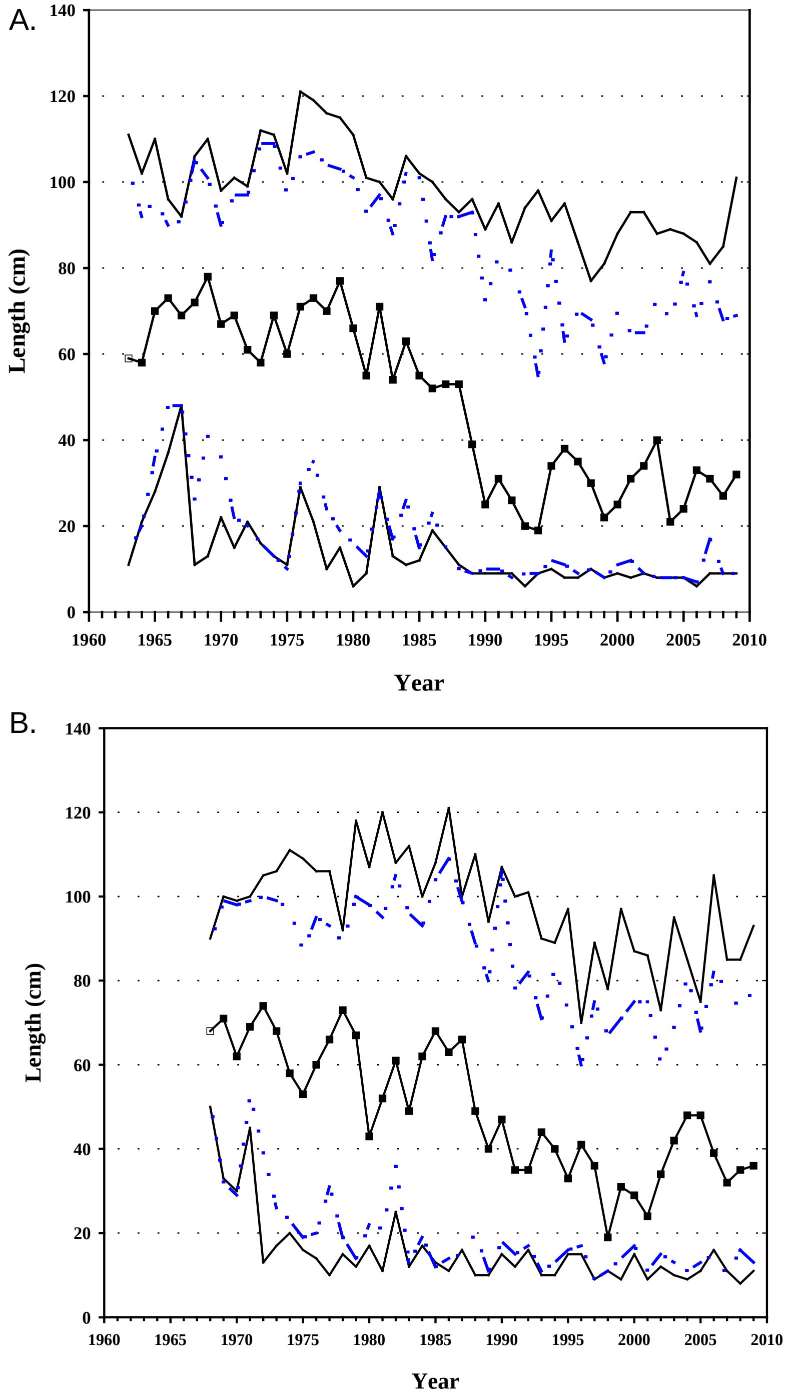


Figure A49. Minimum, median, and, maximum lengths for the northern management region from (A) NEFSC autumn surveys and (B) NEFSC spring surveys

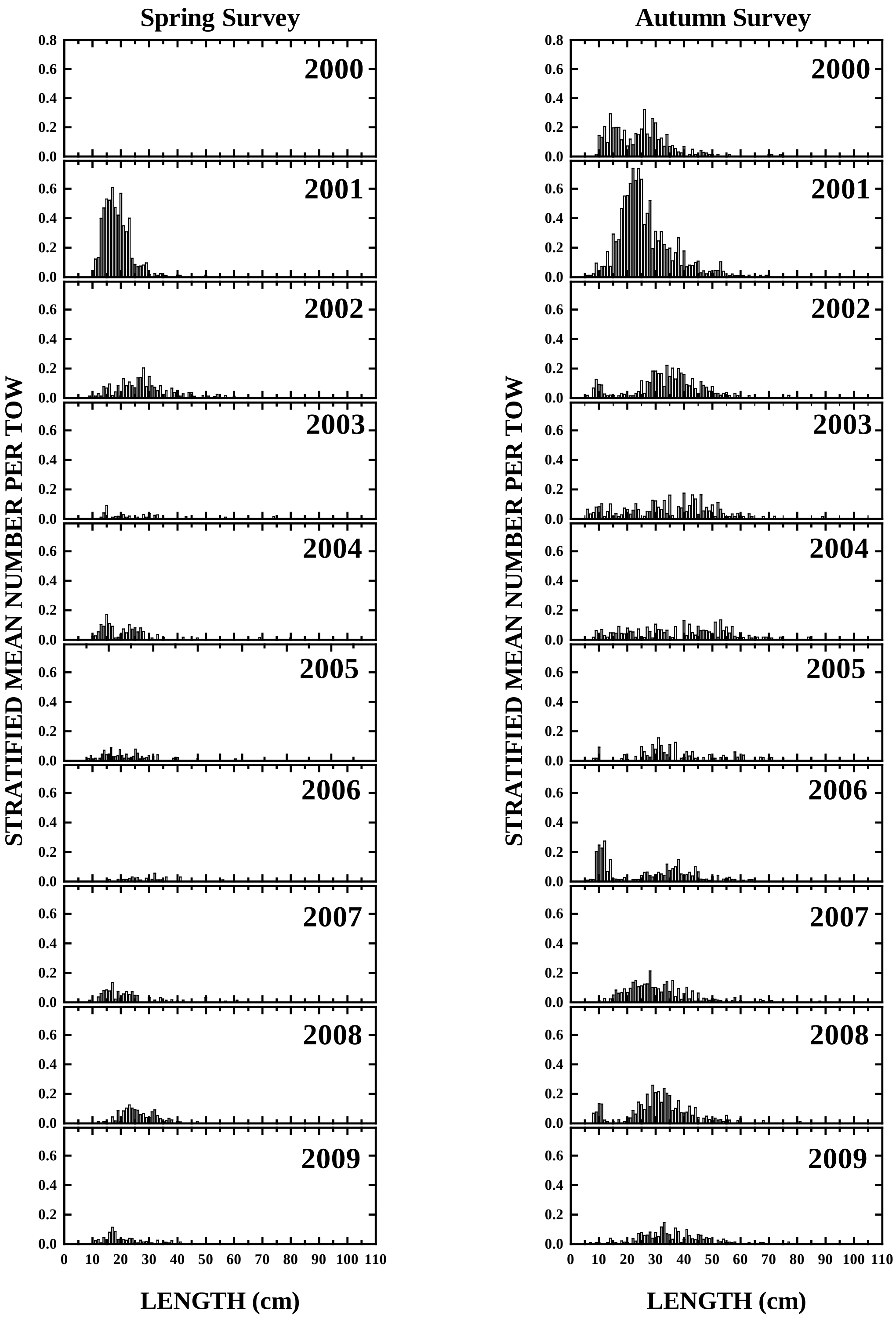
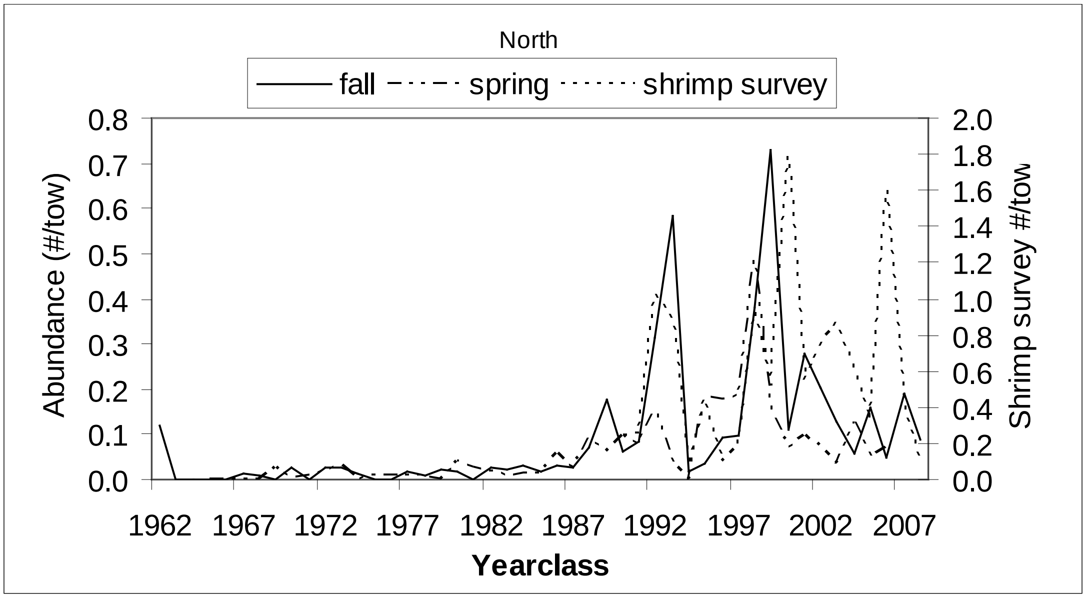


Figure A50. ME-NH inshore survey length frequency plots, 2000-2009.



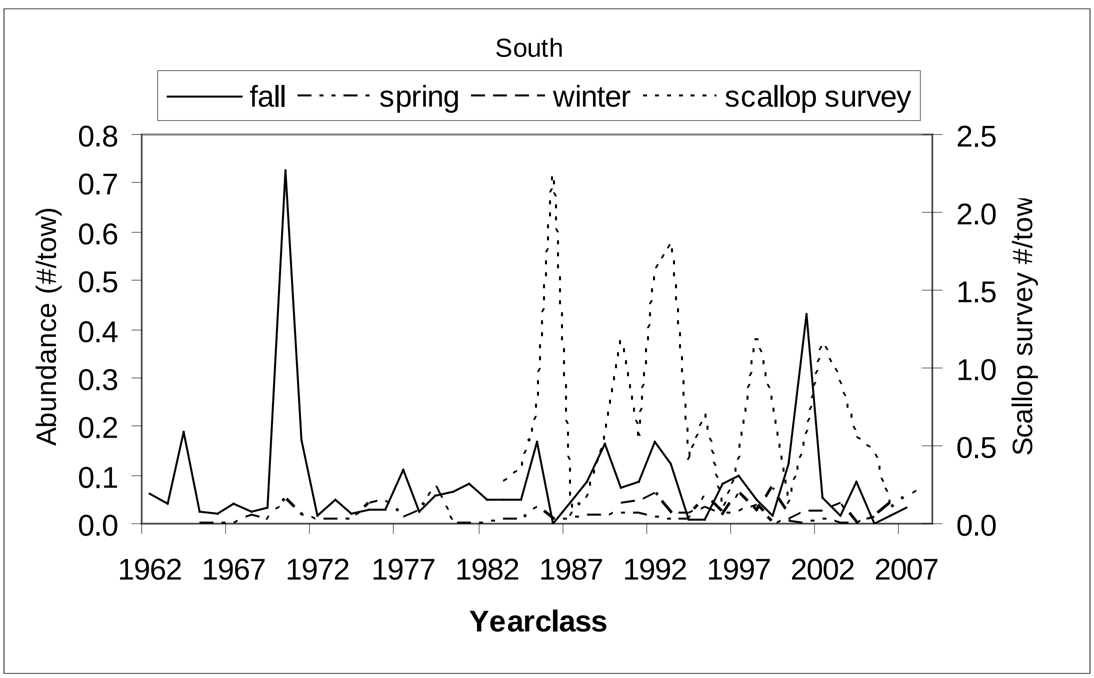
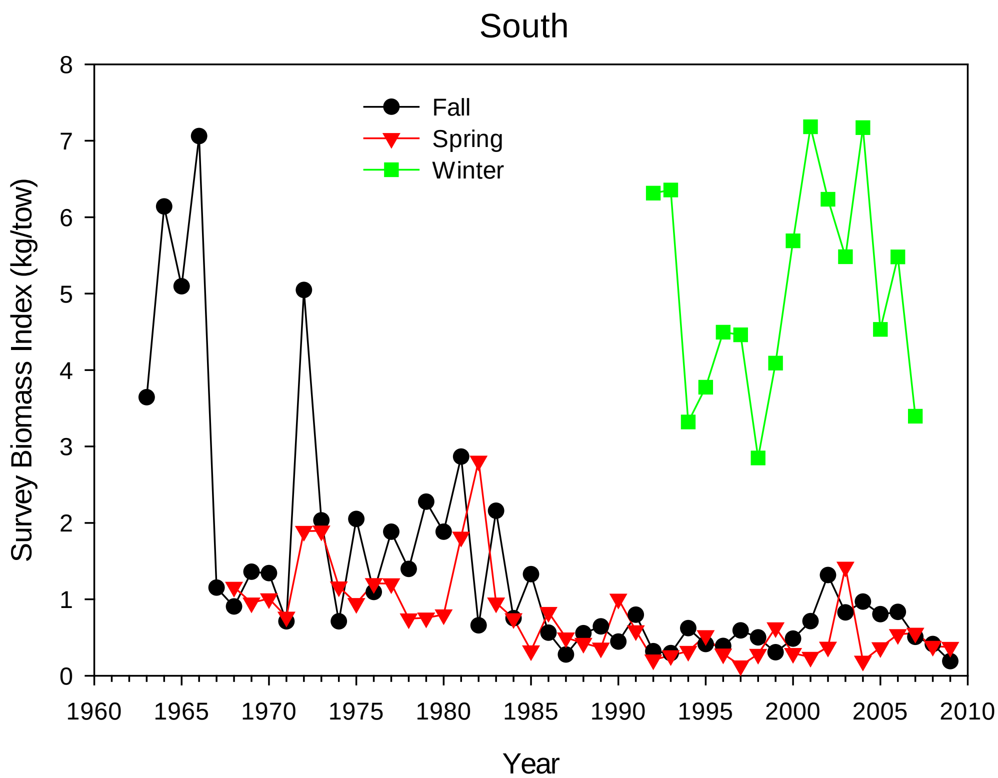


Figure A51. Abundance indices for approximate age 1 (shrimp, scallop and autumn surveys) and age 2 (winter and spring surveys) by yearclass. 2009 FSV Bigelow indices were corrected using calibration coefficient for numbers.



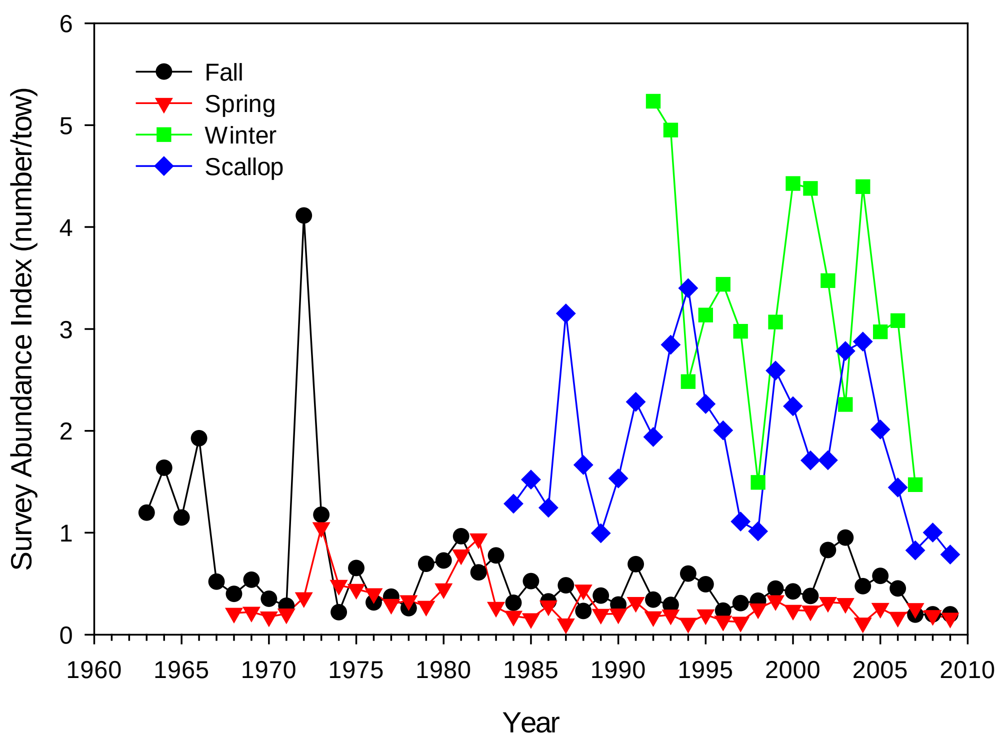


Figure A52. NEFSC spring and autumn surveys of monkfish biomass and abundance in the southern management region.

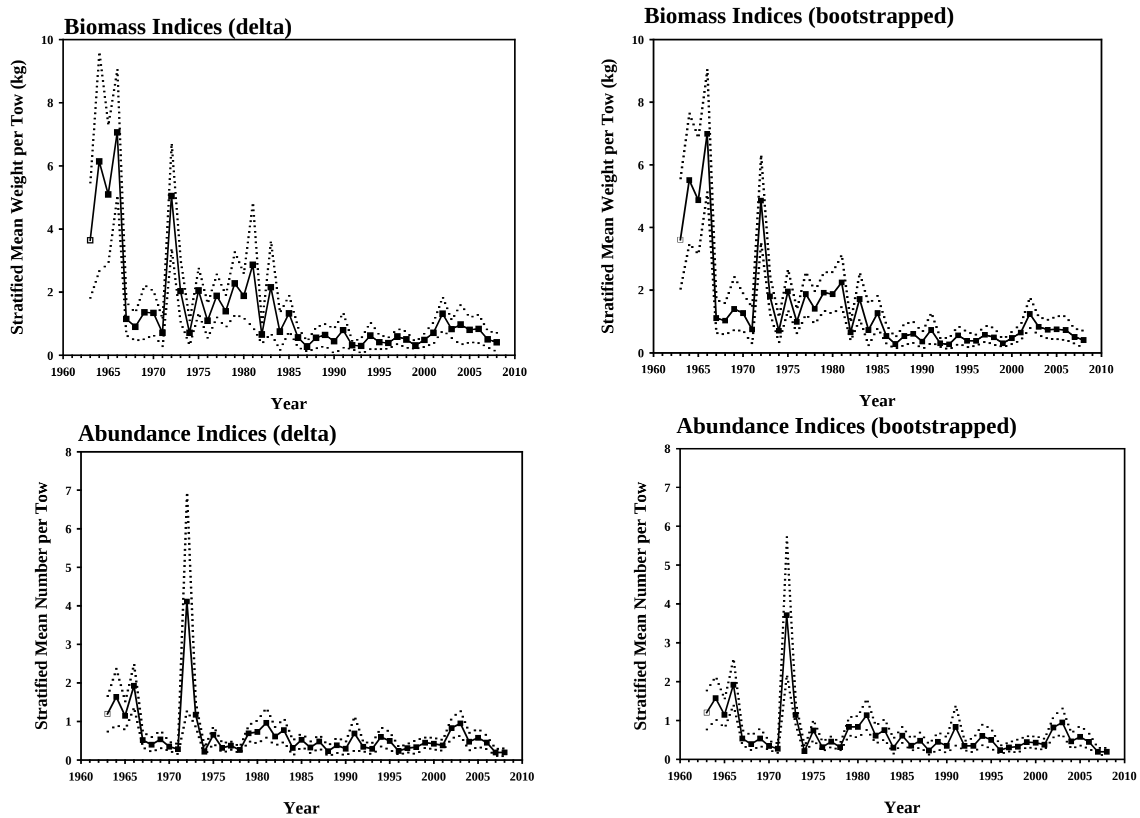


Figure A53. Delta distribution (parametric) and bootstrapped (arithmetic) biomass and abundance indices from the NEFSC autumn bottom trawl survey for the southern management region from 1963-2008. Data prior to 1971 have been revised following an audit of historical data. The 95% confidence limits are shown by the dashed line.

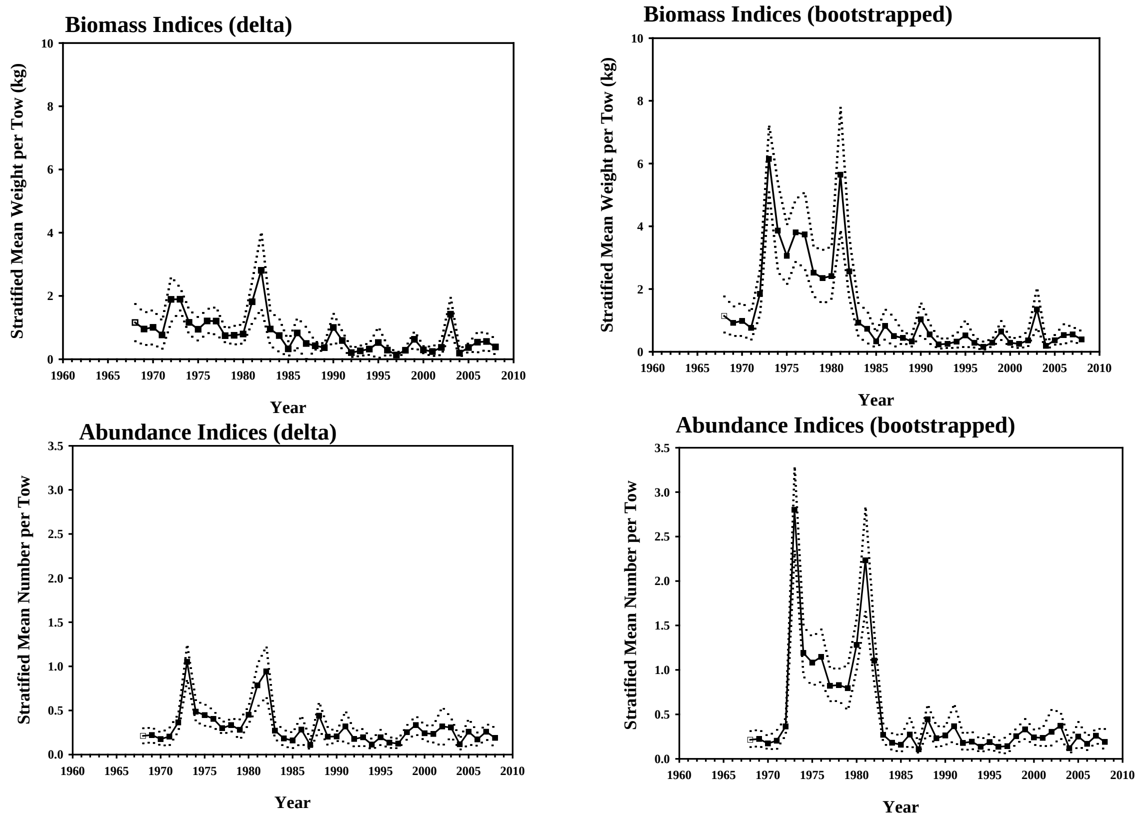


Figure A54. Delta distribution (parametric) and bootstrapped (arithmetic) biomass and abundance indices from the NEFSC spring bottom trawl survey for the southern management region from 1968-2008. Data prior to 1971 have been revised following an audit of historical data. The 95% confidence limits are shown by the dashed line.

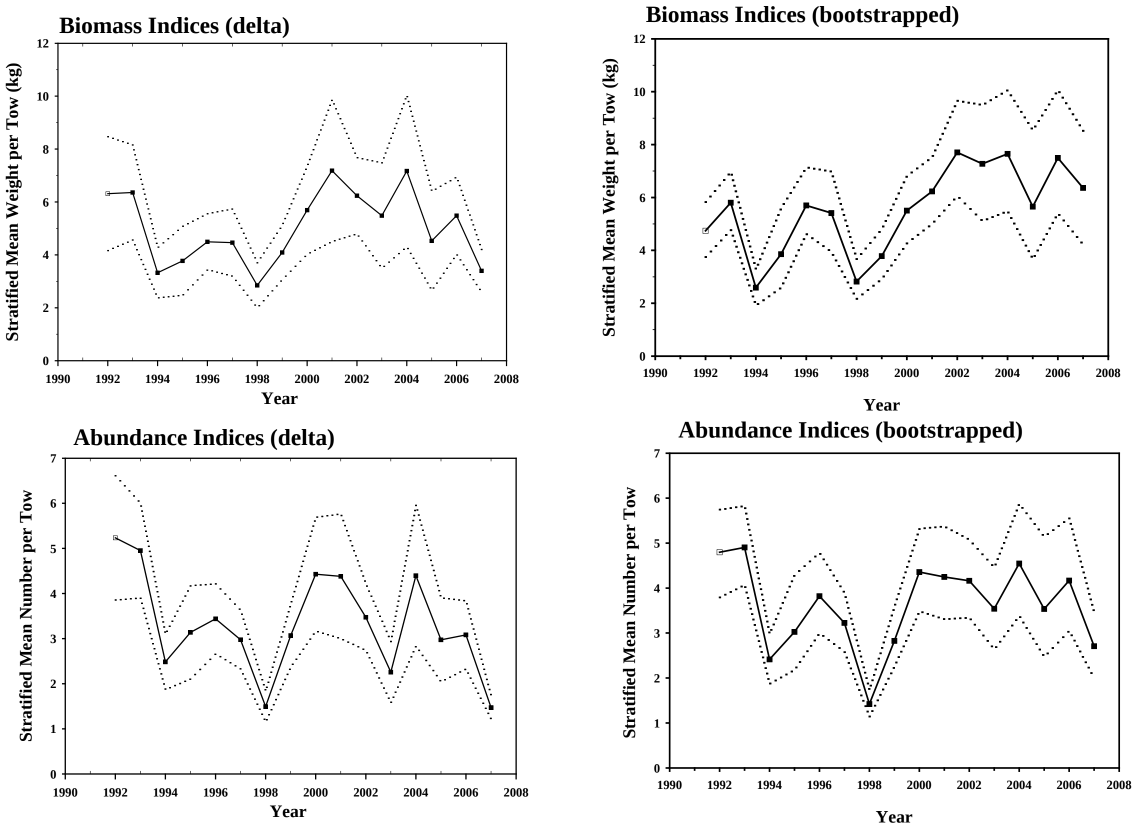


Figure A55. Delta distribution (parametric) and bootstrapped (arithmetic) biomass and abundance indices from the NEFSC winter survey for the southern management region from 1992-2007. Data prior to 1971 have been revised following an audit of historical data. The 95% confidence limits are shown by the dashed line.

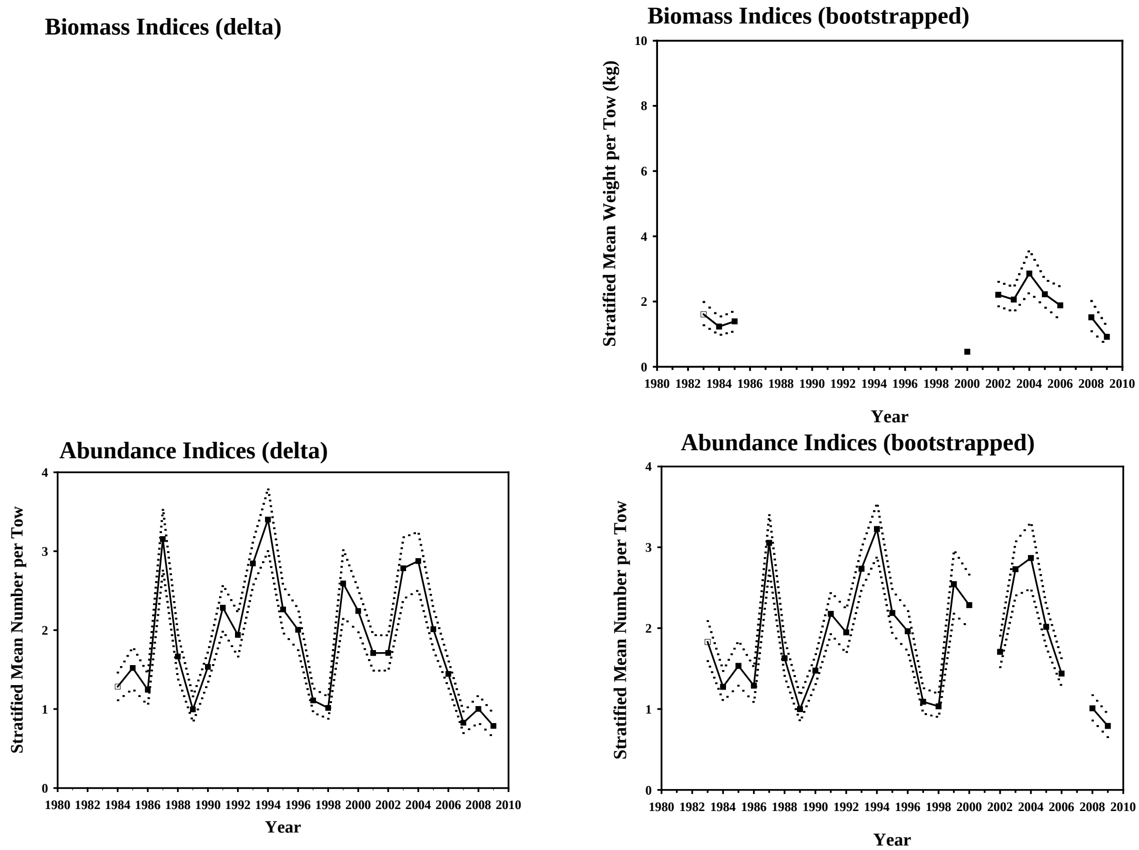


Figure A56. Delta distribution (parametric) and bootstrapped (arithmetic) biomass and abundance indices from the NEFSC scallop survey for the southern management region from 1983-2009. Data prior to 1971 have been revised following an audit of historical data. The 95% confidence limits are shown by the dashed line.

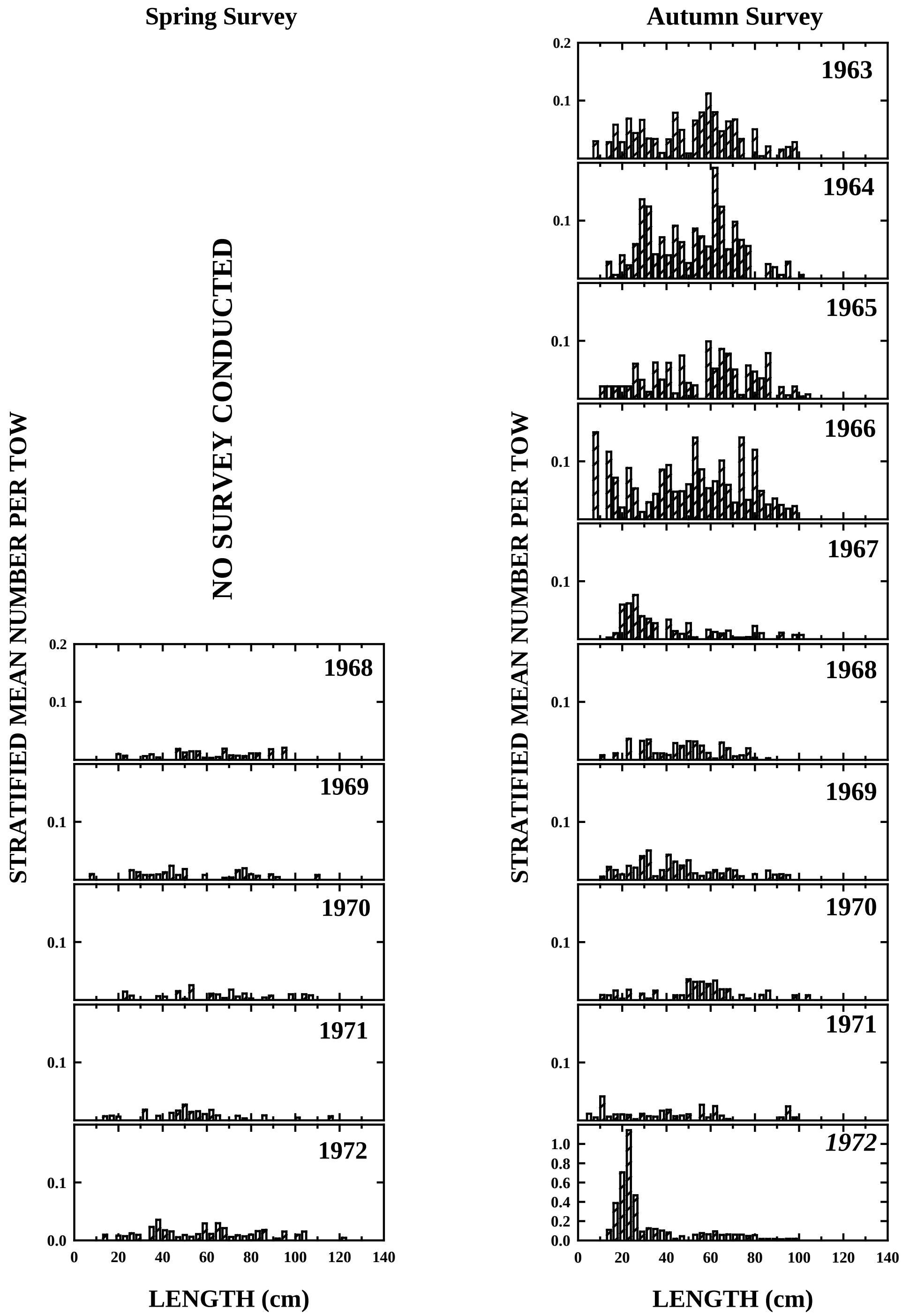


Figure A57, continued

Figure A57. Goosefish length composition from the NEFSC spring bottom trawl (March-April), winter flatfish (February), summer scallop (July-August), and autumn (September-October) bottom trawl surveys in the southern management region, 1963-2009. Note: 1963-1966 sampled reduced strata set

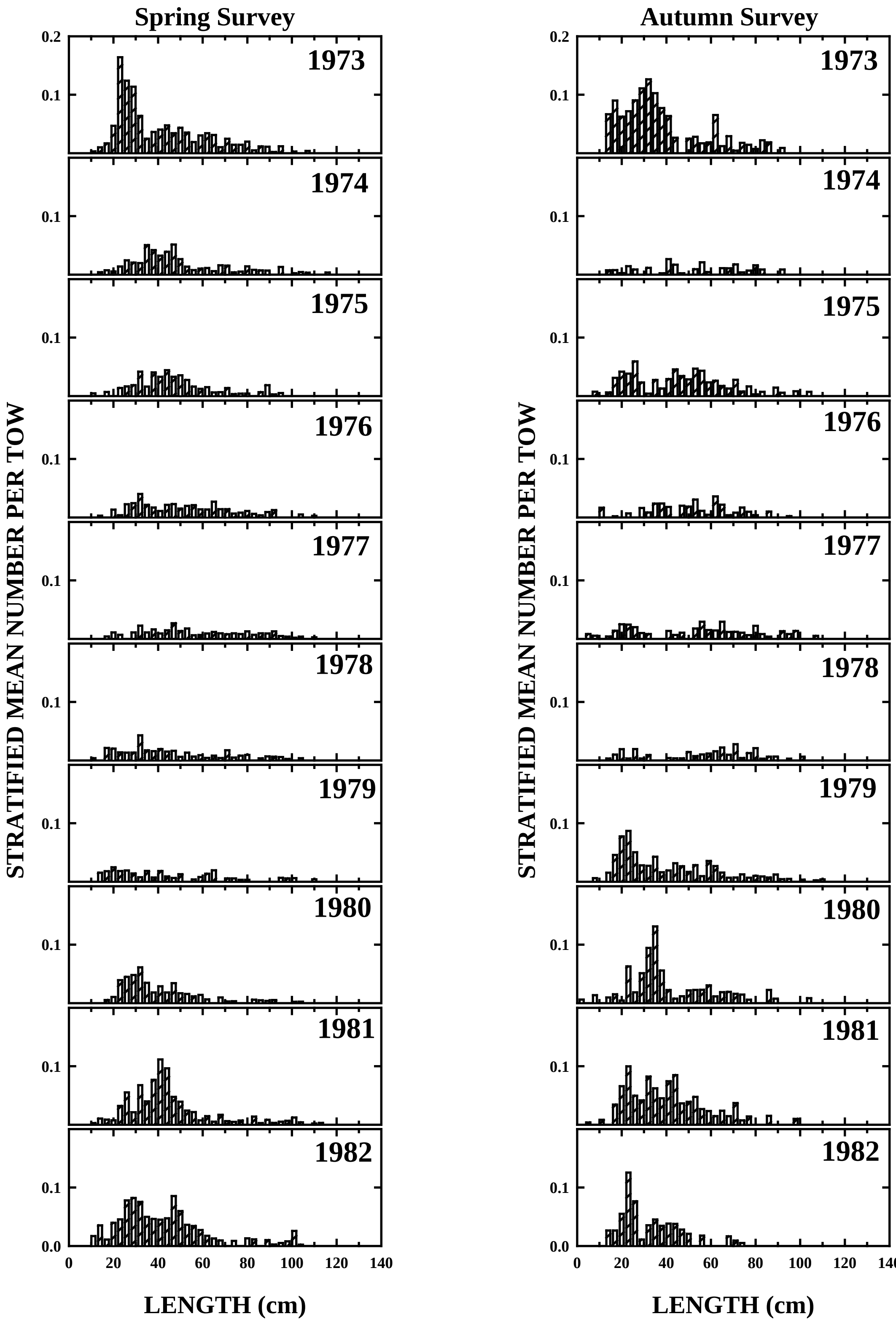


Figure A57, continued

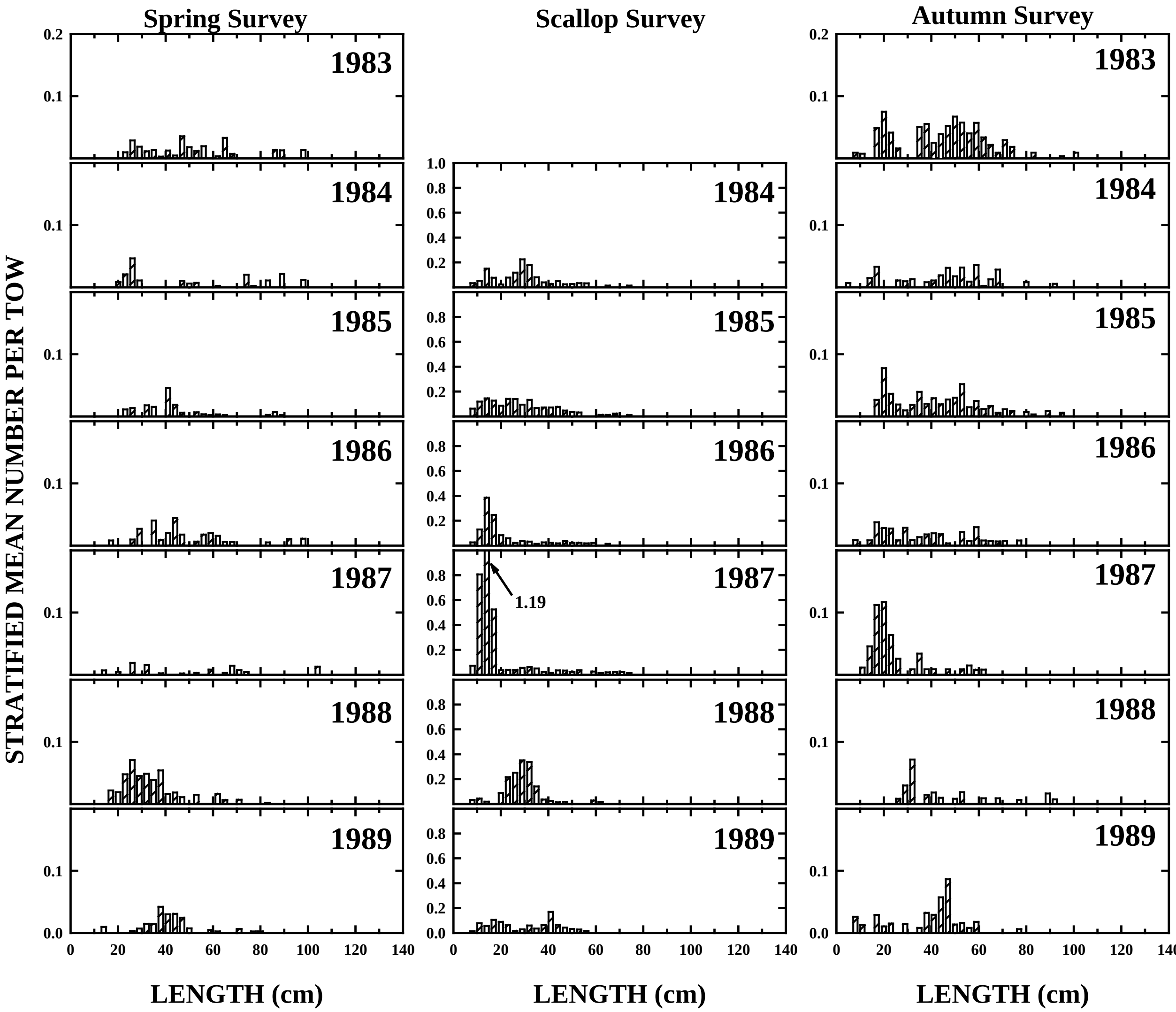


Figure A57, continued

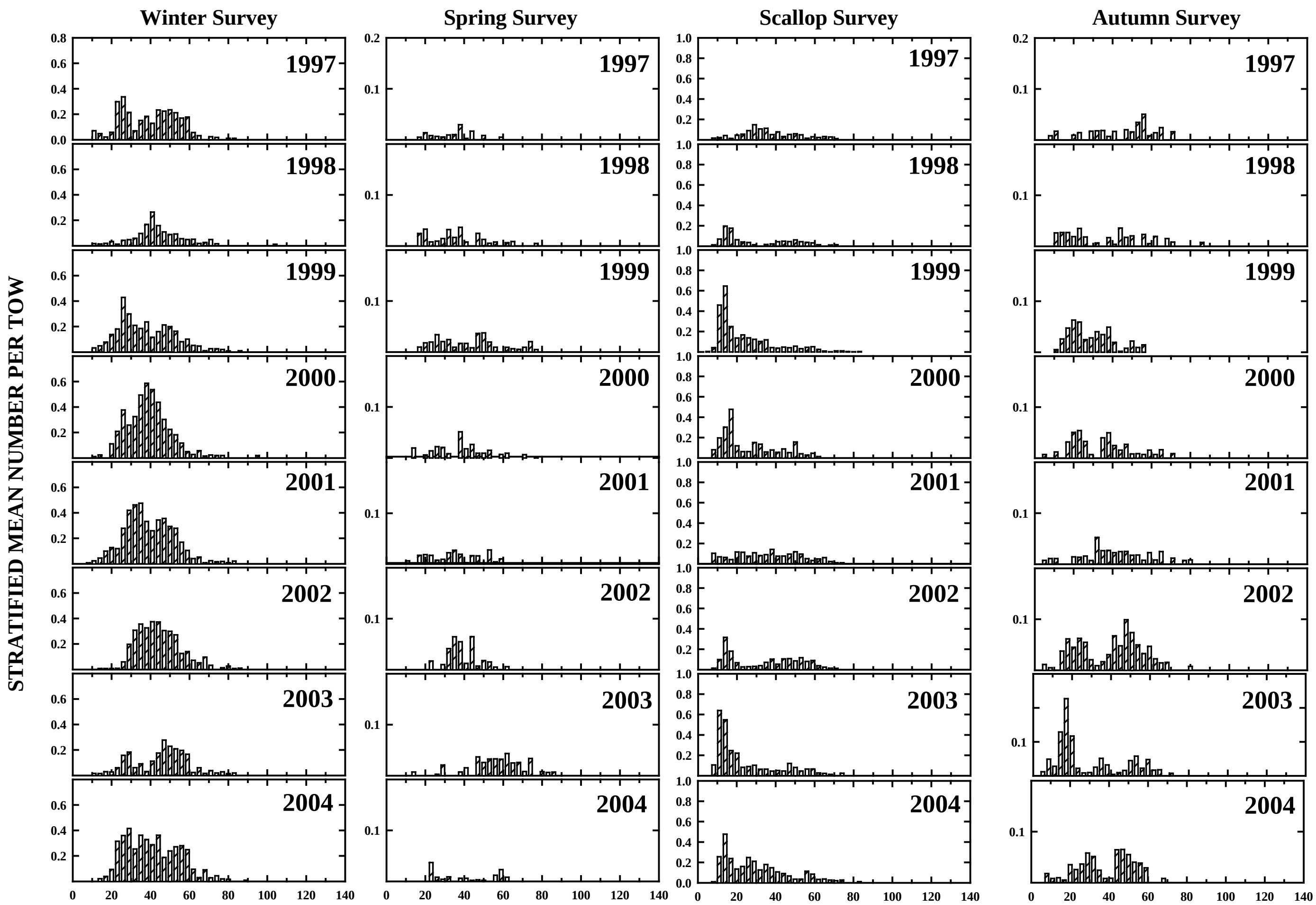


Figure A57, continued

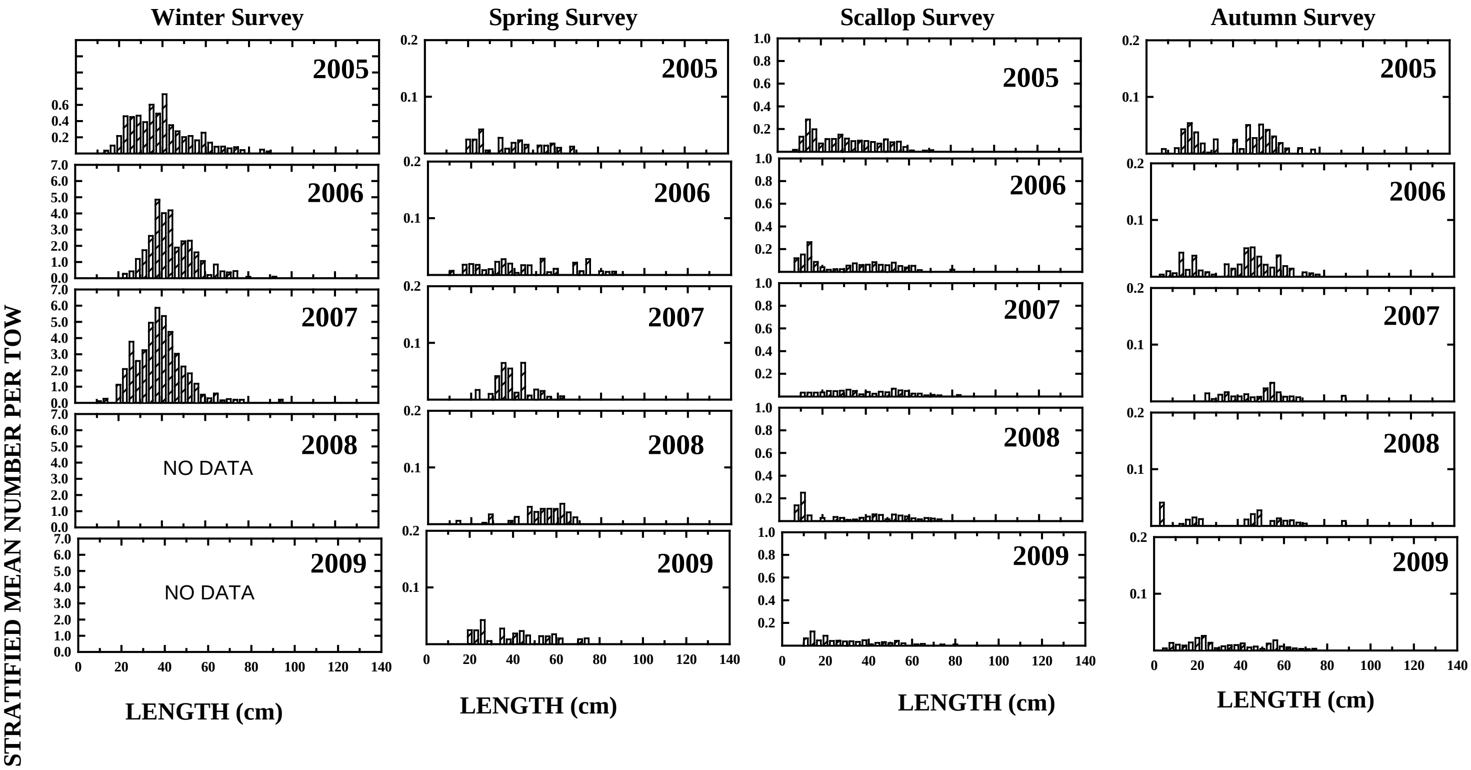
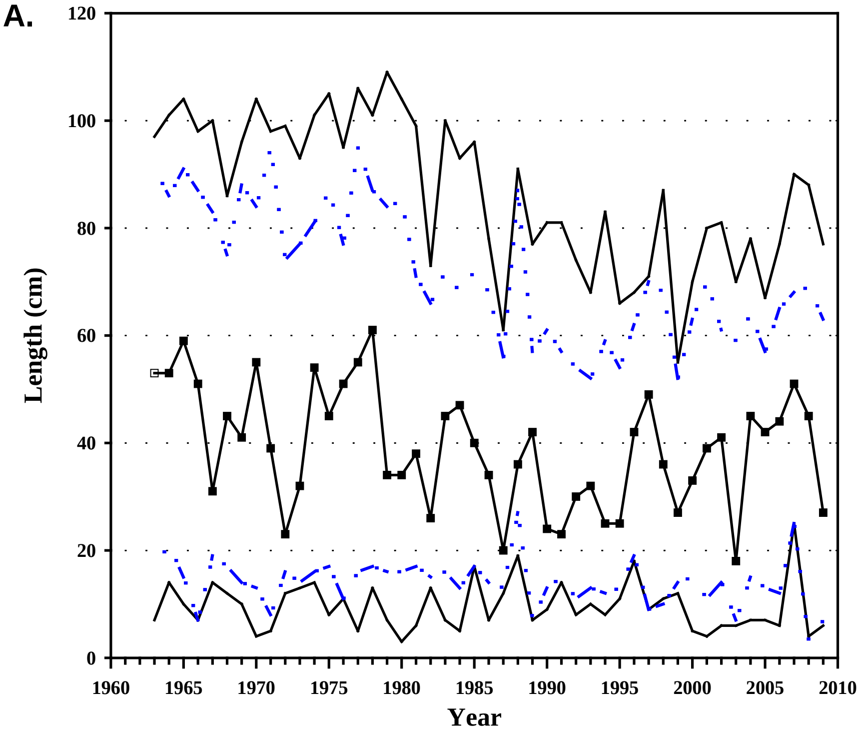


Figure A57, continued

A.



B.

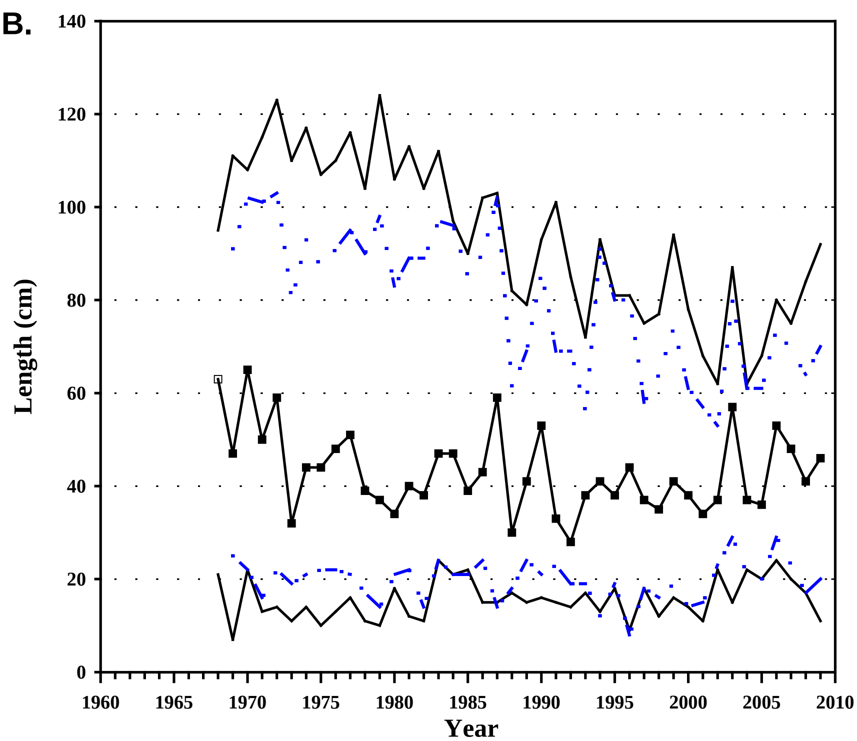


Figure A58. Minimum, median, and, maximum lengths for the southern management region from (A) NEFSC autumn surveys and (B) NEFSC spring surveys.

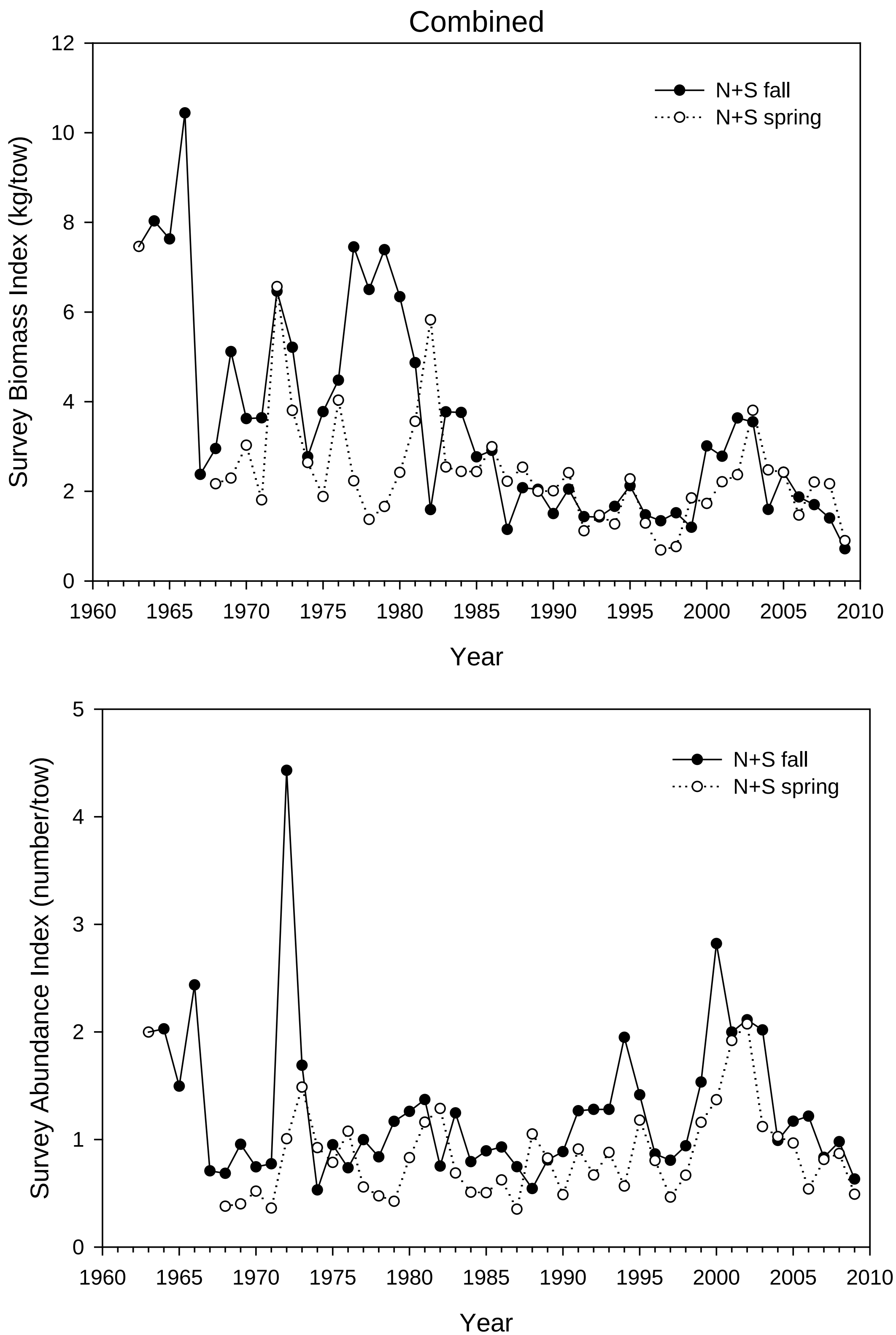


Figure A59. NEFSC spring and autumn surveys of monkfish biomass and abundance in both the northern and southern management regions

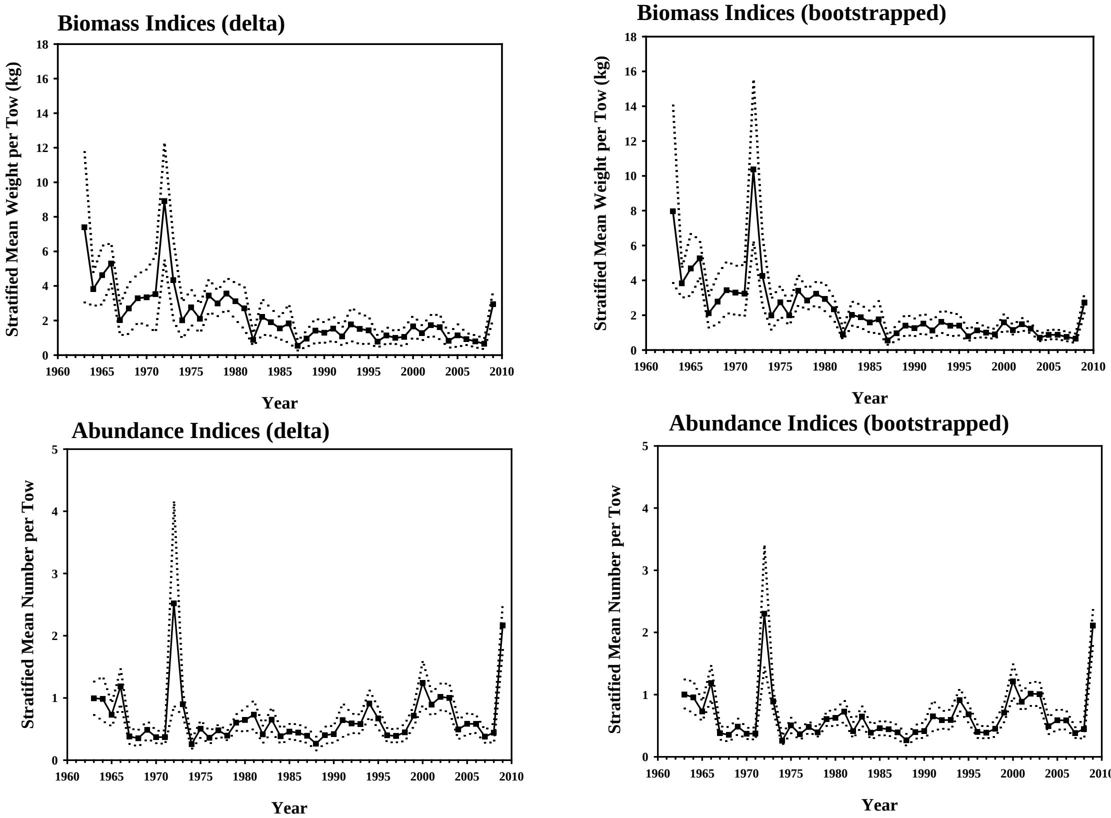


Figure A60. Delta distribution (parametric) and bootstrapped (arithmetic) biomass and abundance indices from the NEFSC autumn bottom trawl survey for combined management regions from 1963-2009. Data prior to 1971 have been revised following an audit of historical data. The 95% confidence limits are shown by the dashed line.

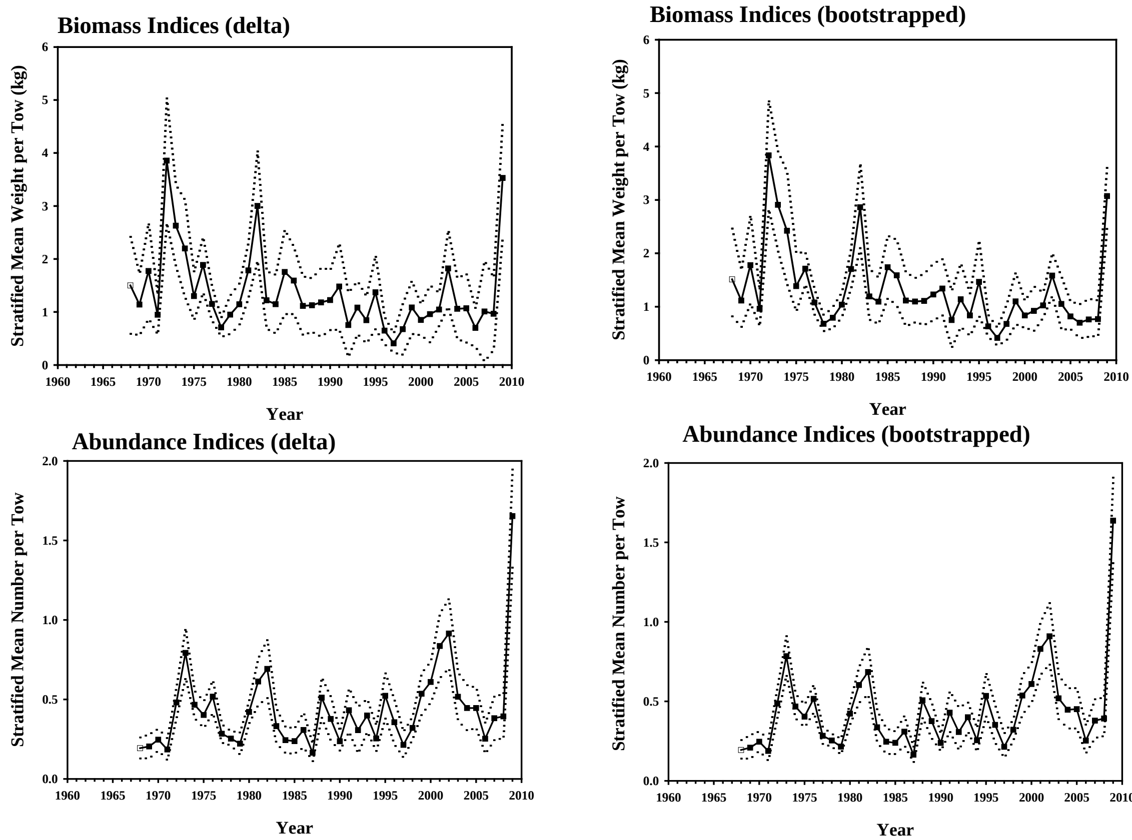


Figure A61. Delta distribution (parametric) and bootstrapped (arithmetic) biomass and abundance indices from the NEFSC spring bottom trawl survey for management regions combined from 1963-2009. Data prior to 1971 have been revised following an audit of historical data. The 95% confidence limits are shown by the dashed line.

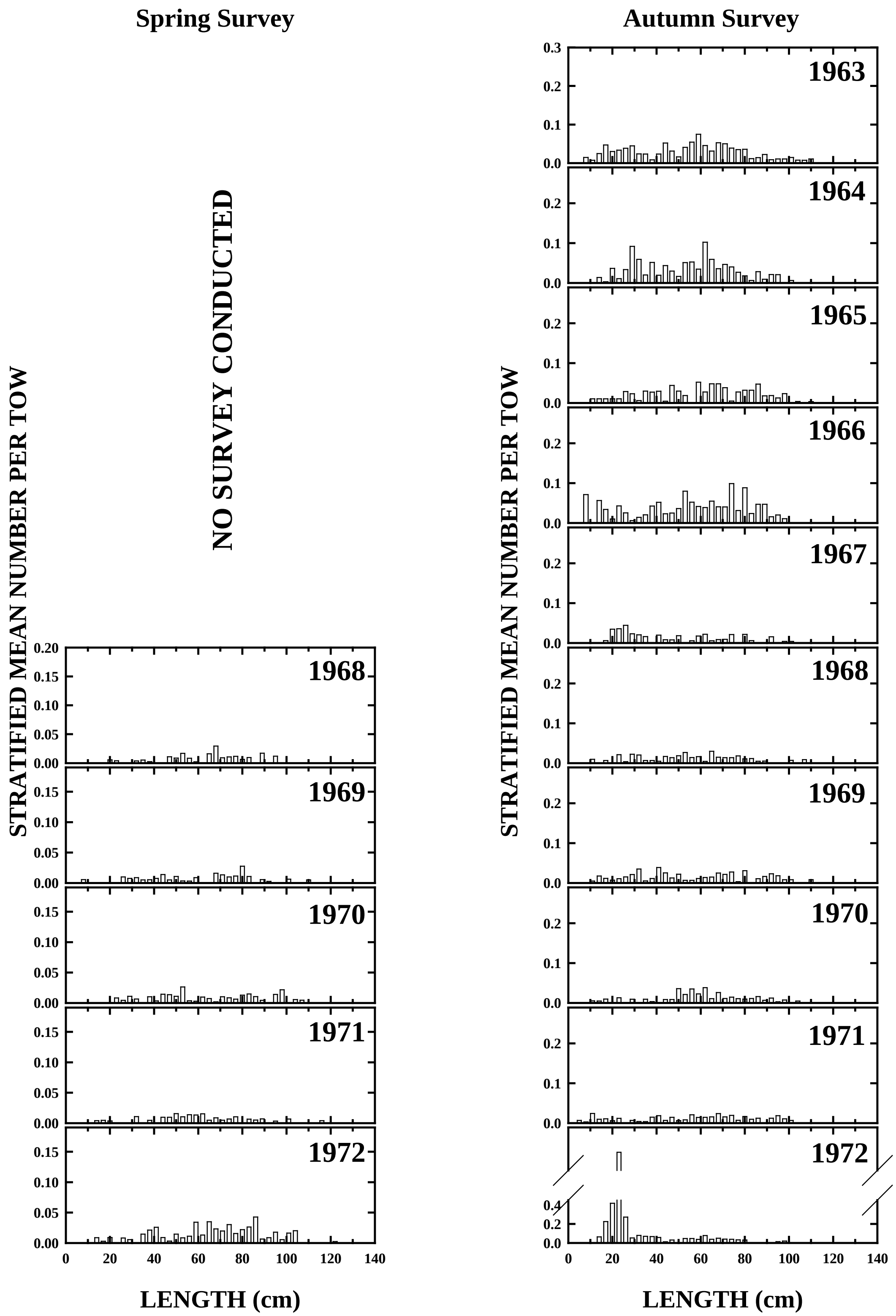


Figure A62. Goosefish length composition from the NEFSC spring and autumn bottom trawl surveys in both management regions combined, 1963-2009.

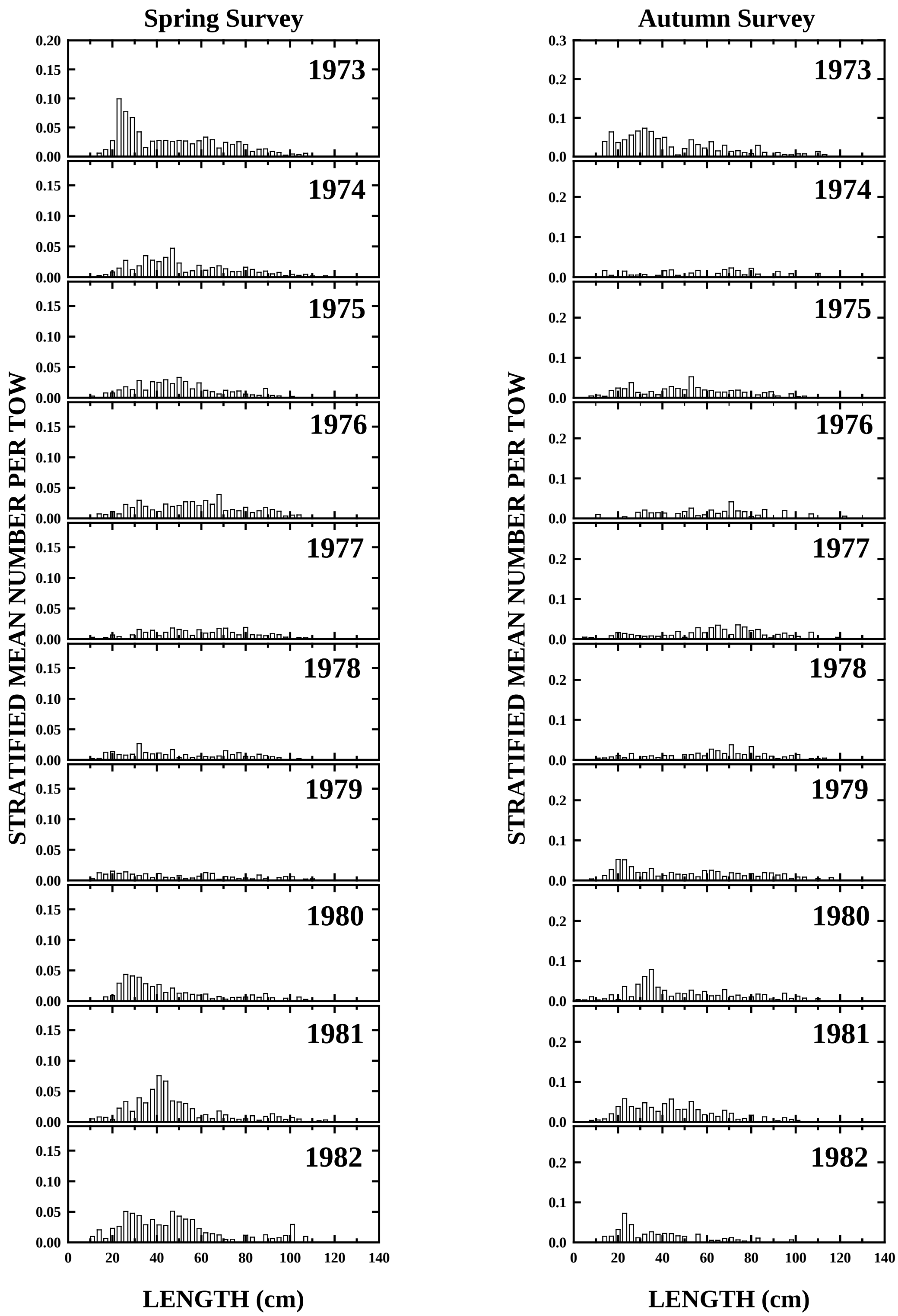


Figure A62, continued

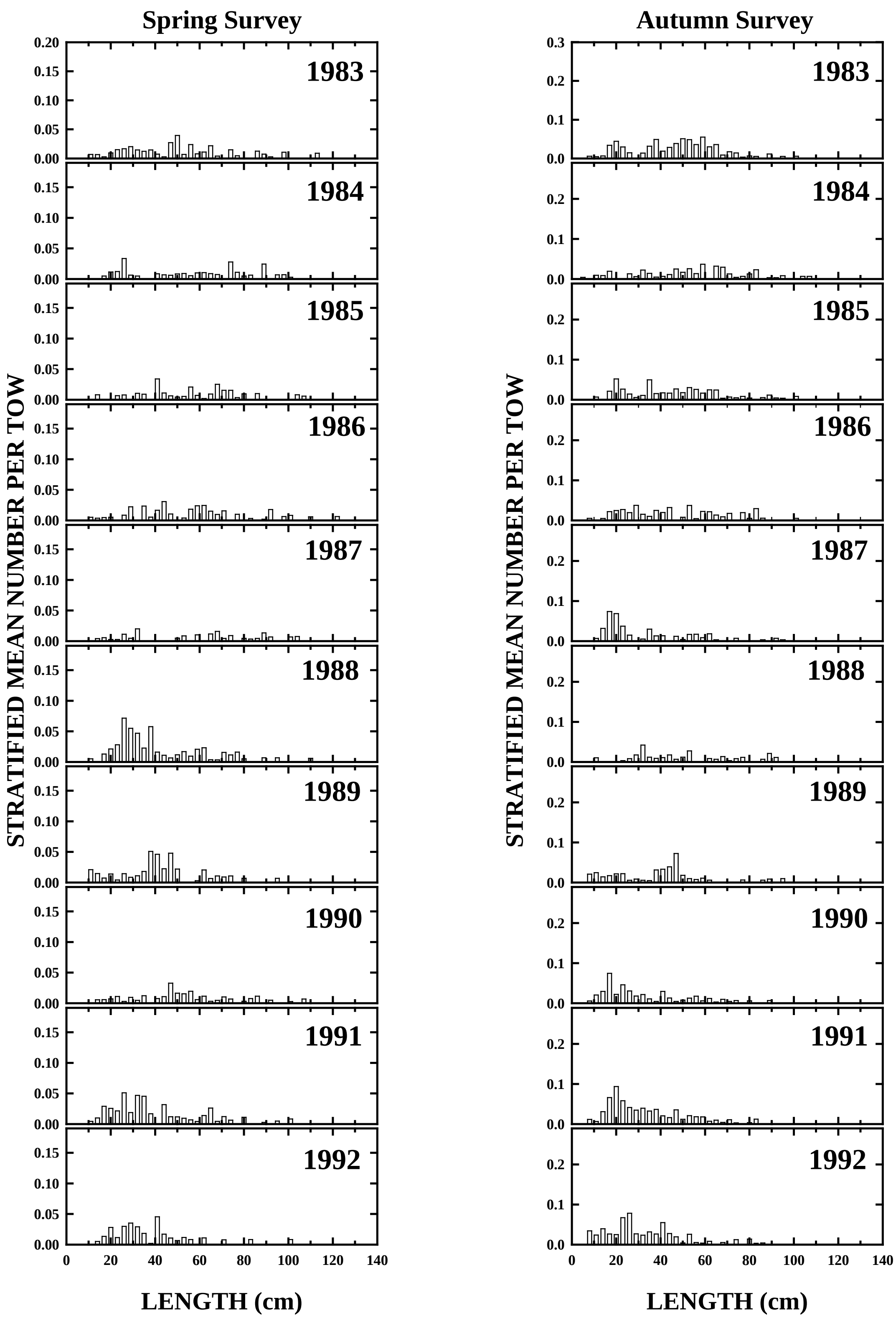
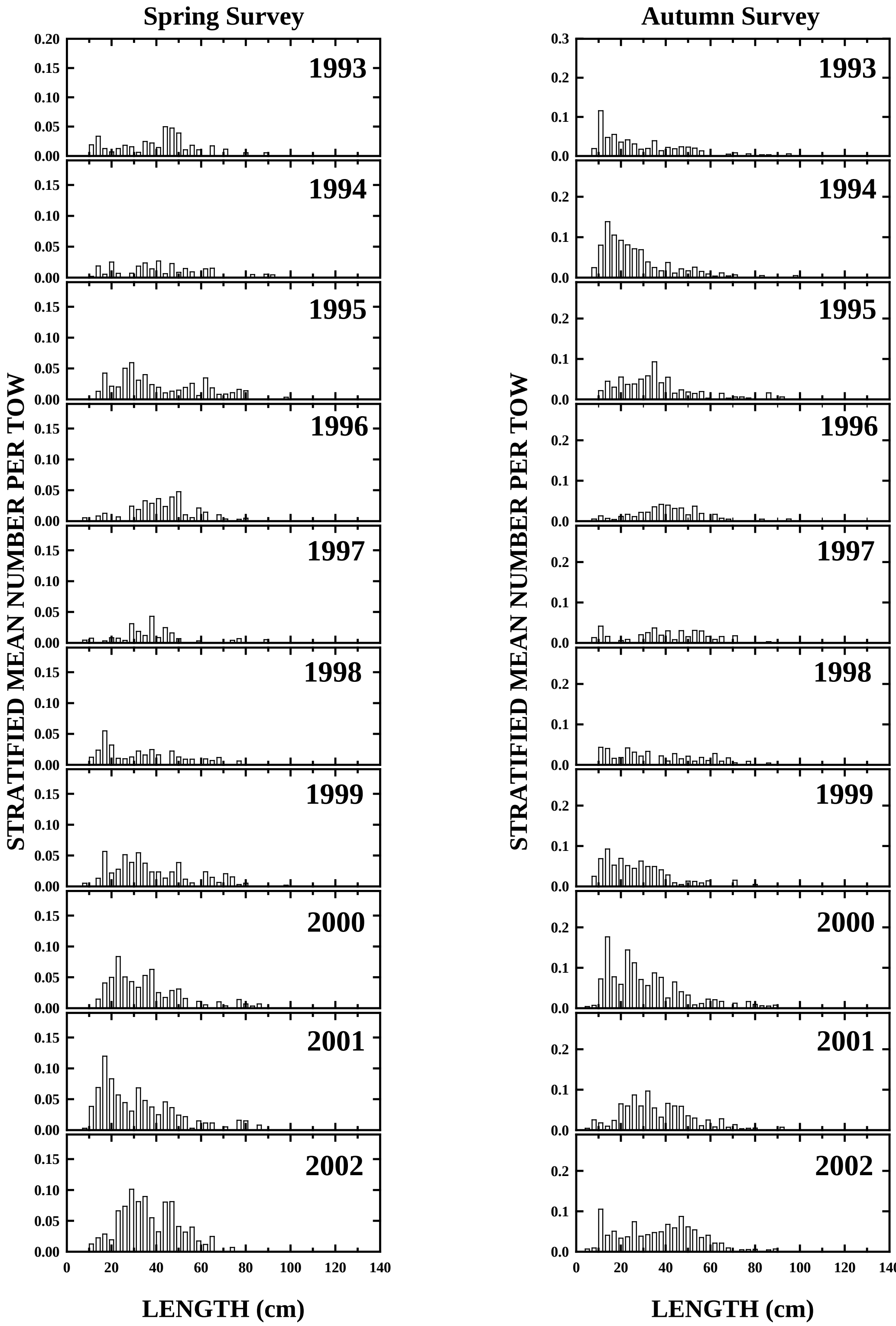


Figure A62, continued



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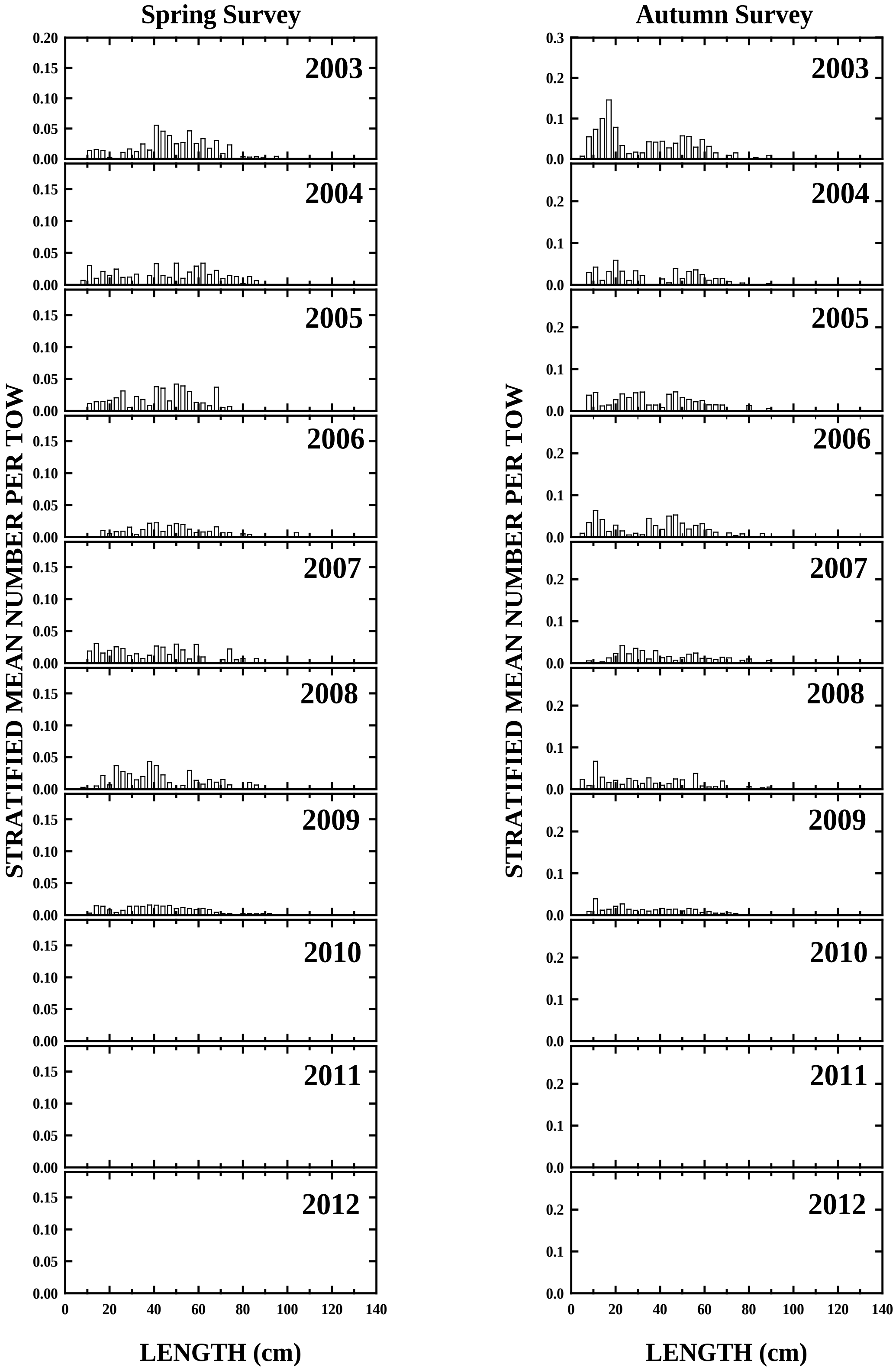

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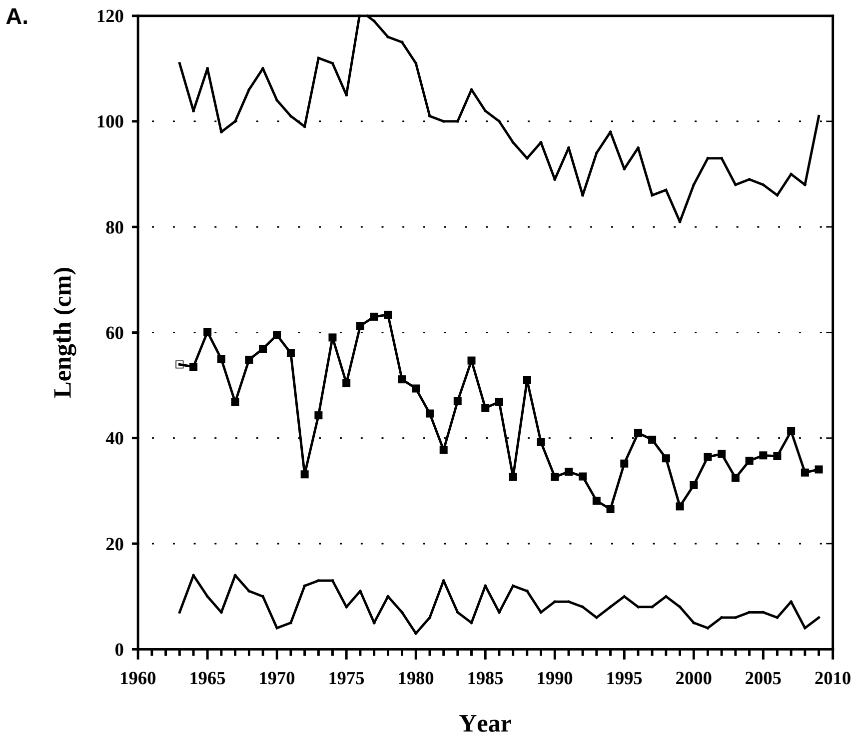
Figure A62, continued



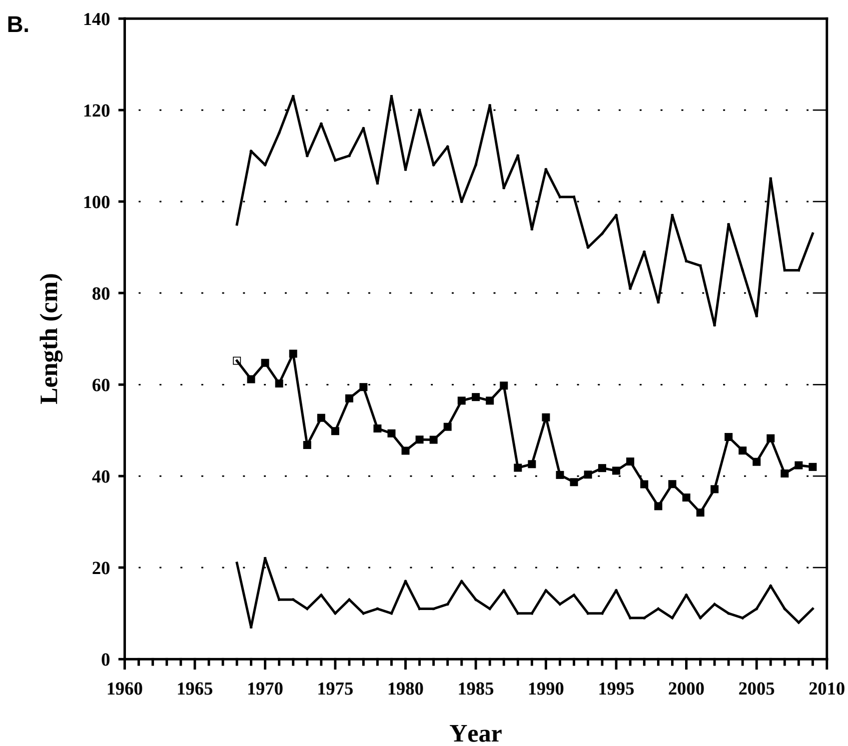


Figure A63. Minimum, median, and, maximum lengths for the northern and southern management regions combined from (A) NEFSC autumn surveys and (B) NEFSC spring surveys.

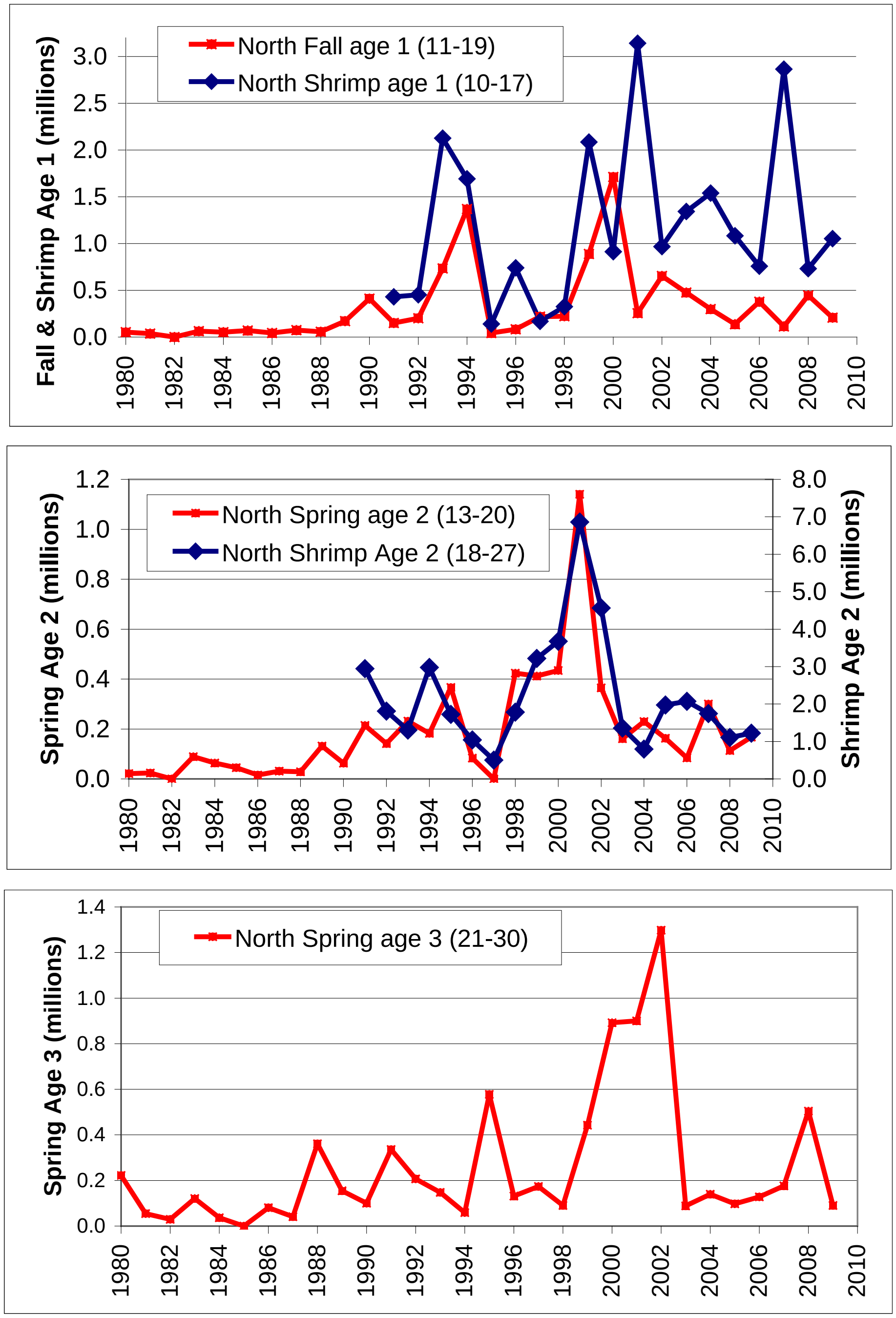


Figure A64. Northern management area monkfish recruitment indices at age. Centimeter intervals used to estimate recruitment ages are given in parenthesis.

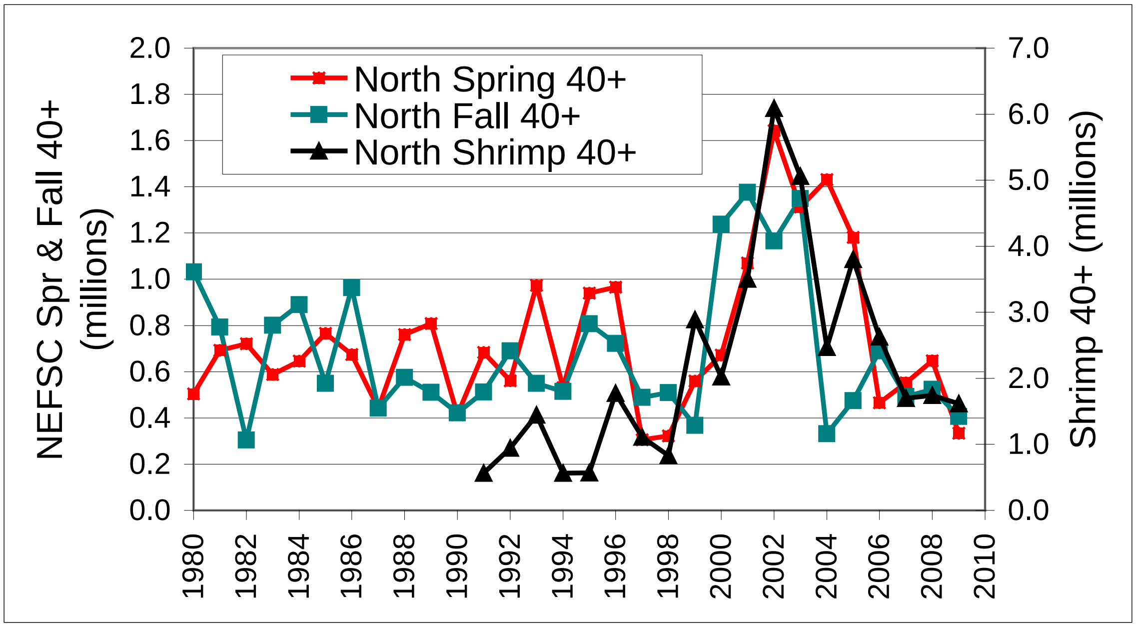
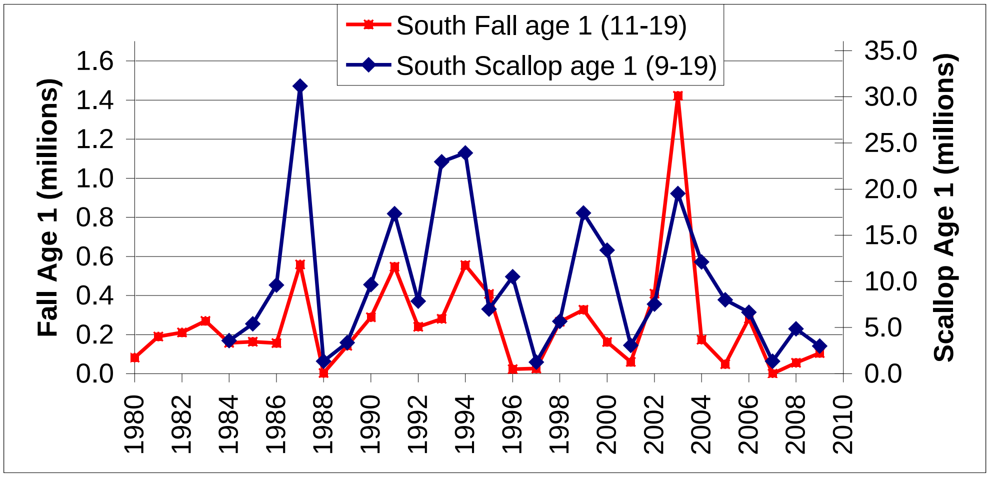
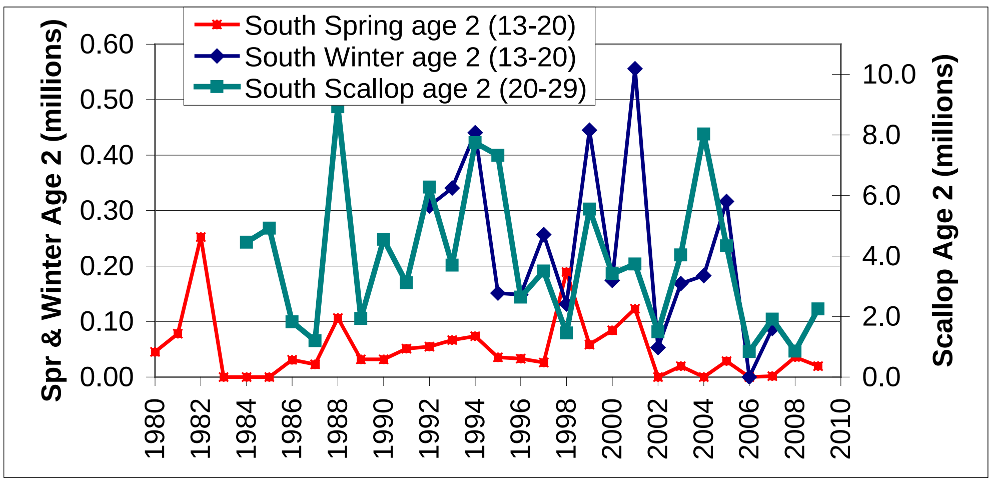


Figure A65. Adult 40+ cm abundance indices for the northern management area.





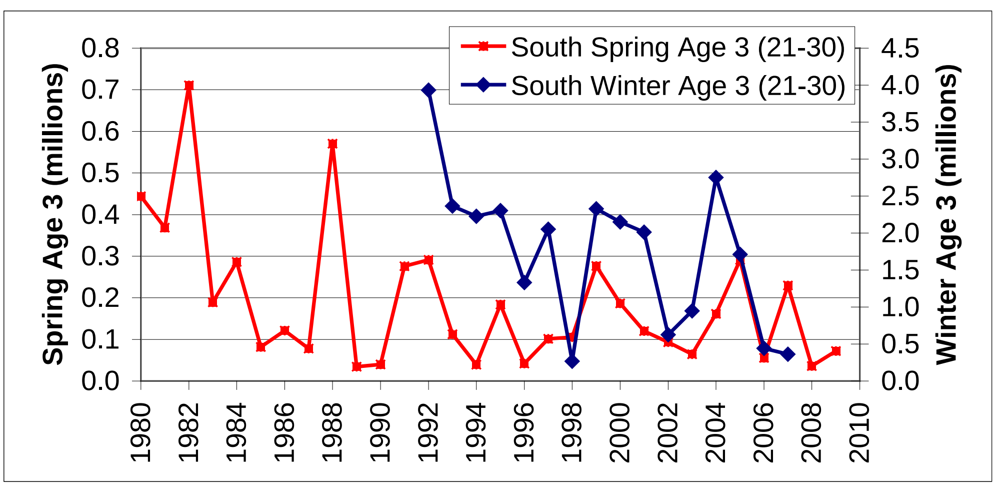


Figure A66. Southern management area monkfish recruitment indices at age. Centimeter interval used to estimate recruitment ages are given in parenthesis.

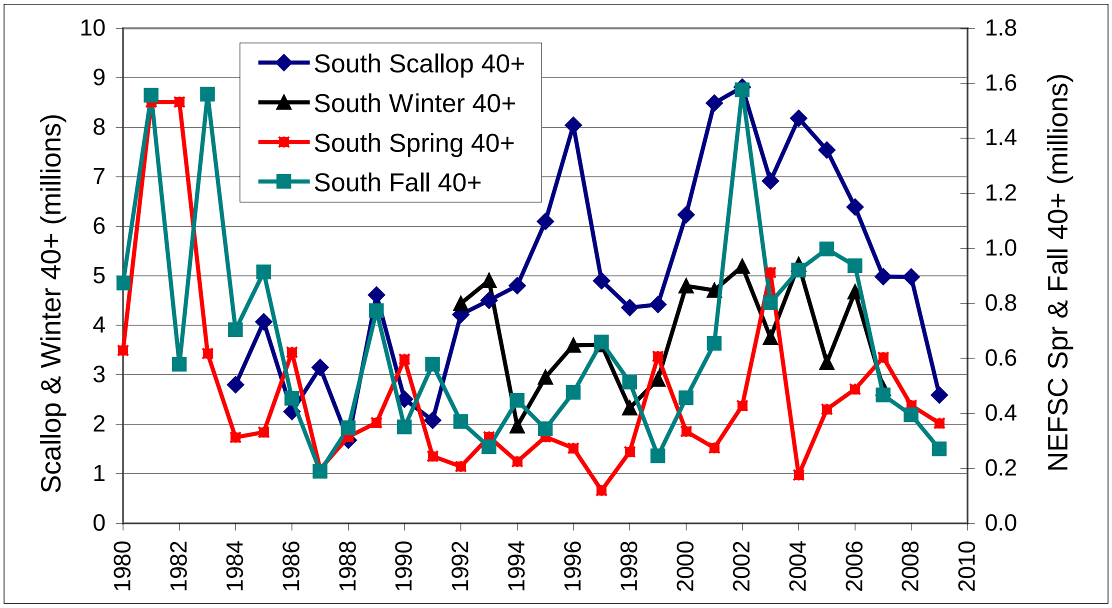


Figure A67. Adult 40+ cm abundance indices for the southern management area.

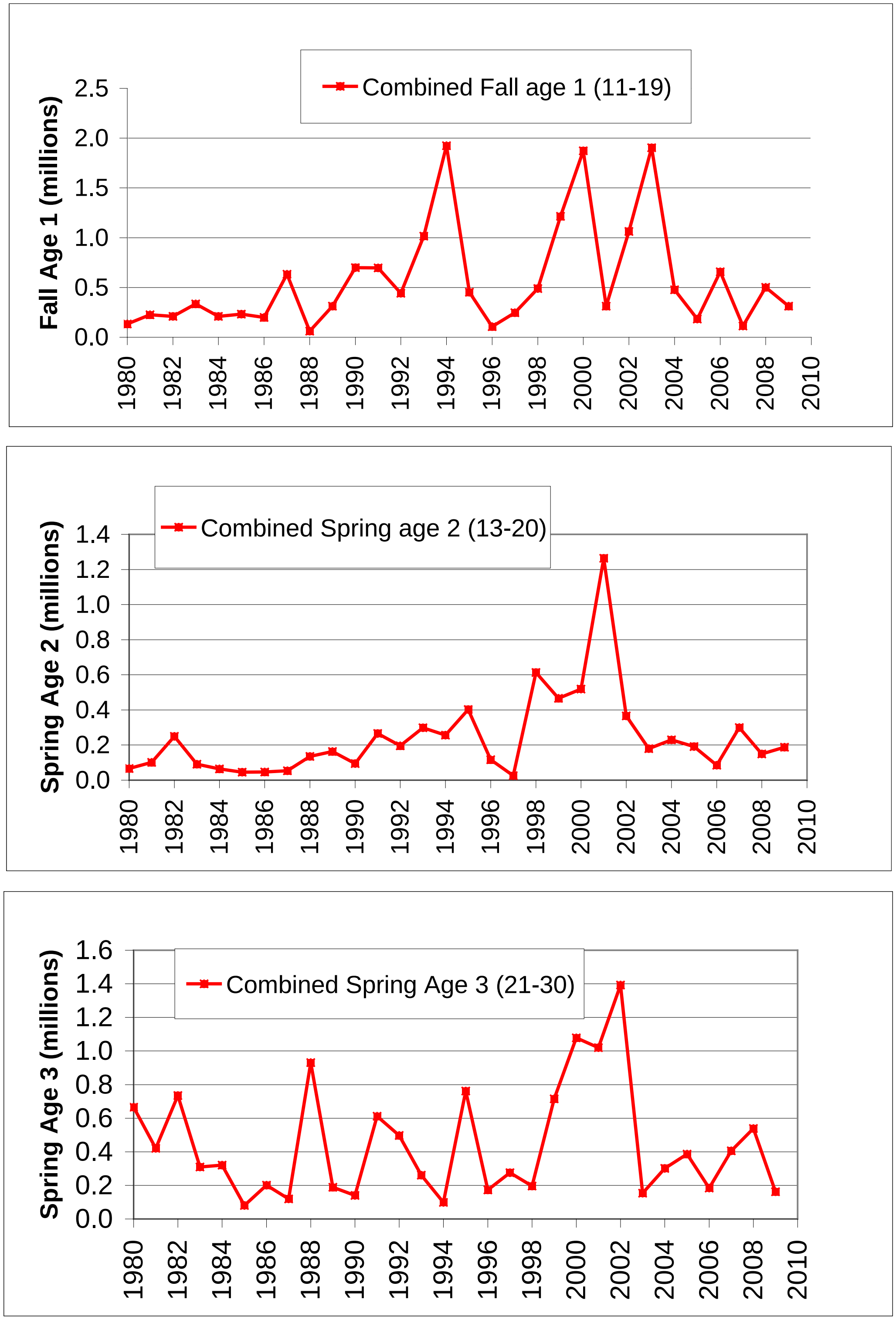


Figure A68. Combined management areas monkfish recruitment indices at age for the NEFSC surveys. Centimeter intervals used to estimate recruitment ages are given in parentheses.

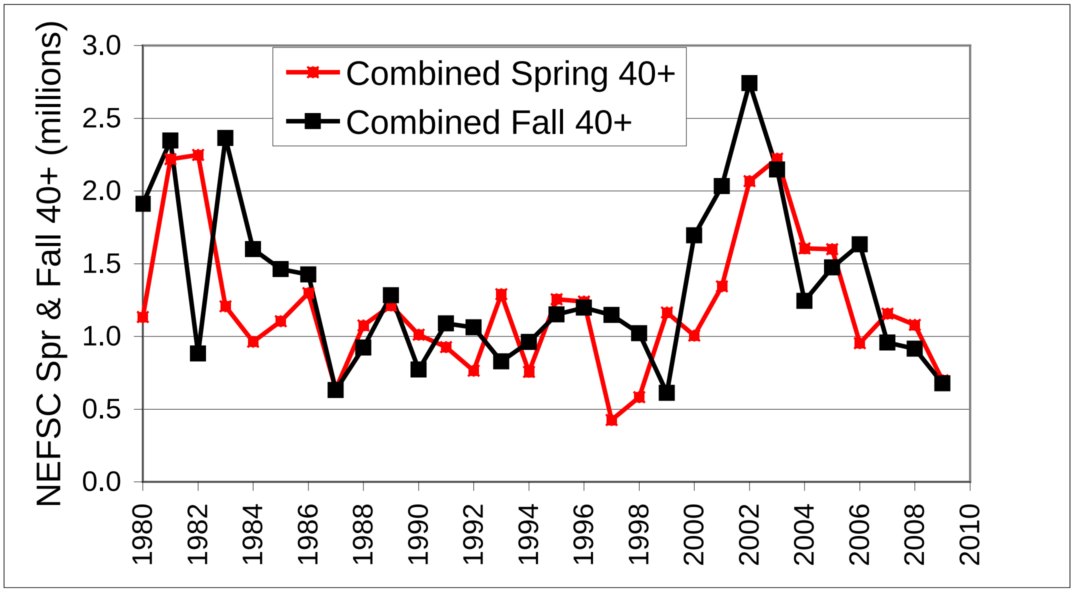


Figure A69. Adult 40+ cm abundance indices for the combined management areas for the NEFSC bottom trawl surveys

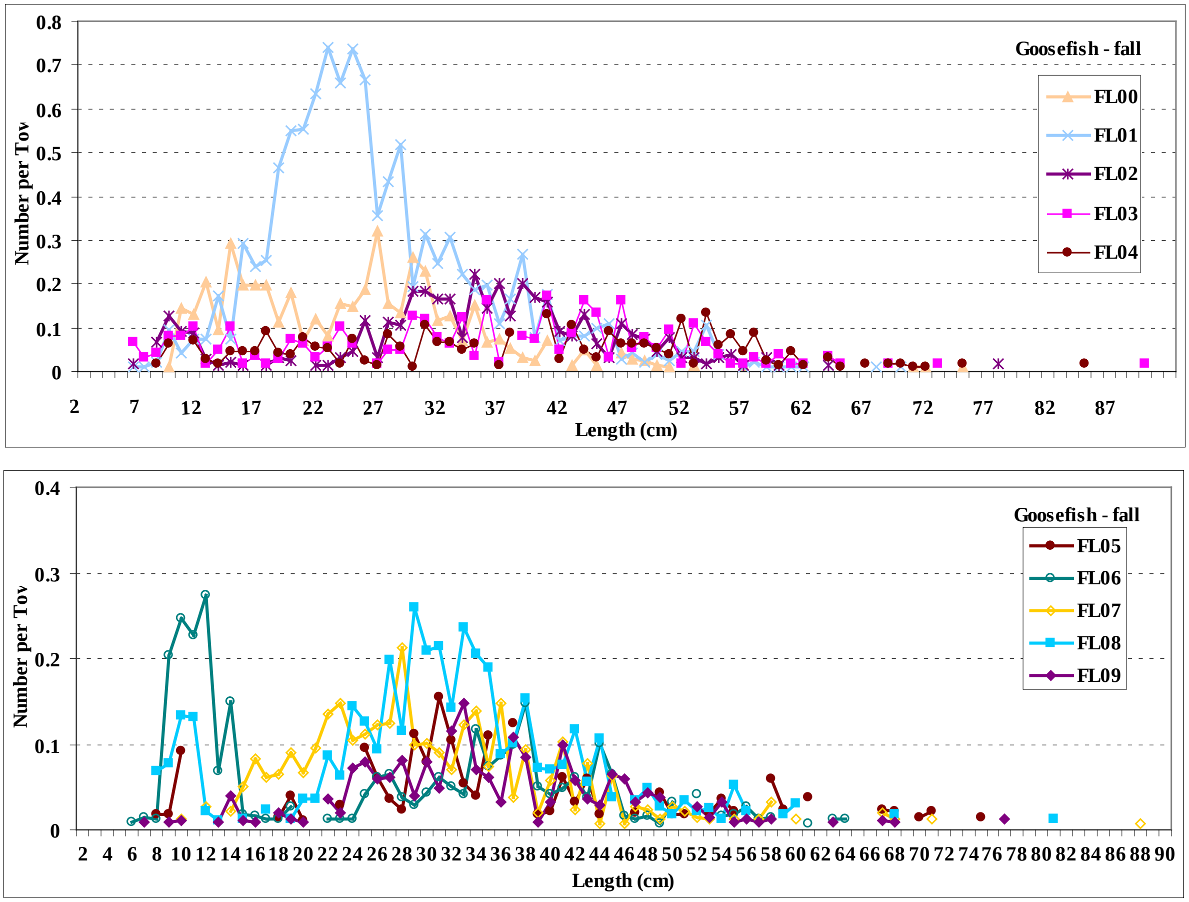


Figure A70. Length frequency distributions from the fall ME/NH bottom trawl survey from 2000 to 2009.

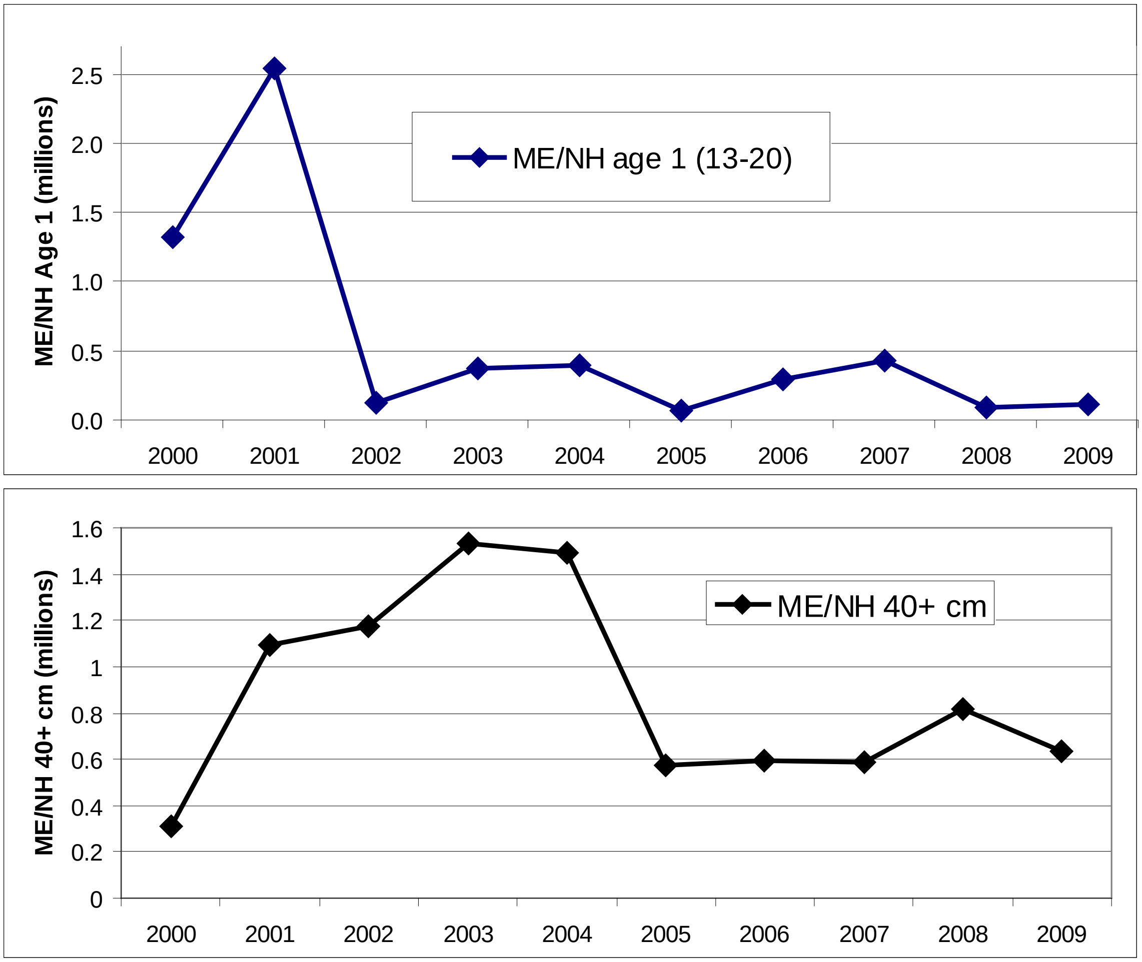
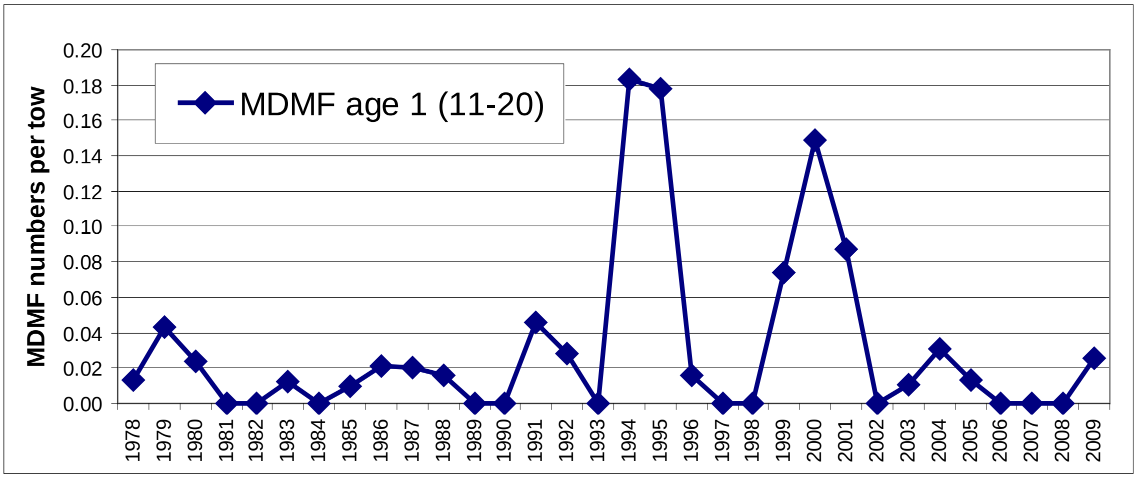


Figure A71. Age 1 (13 to 20cm) and 40+ cm indices from the fall ME/NH bottom trawl survey from 2000 to 2009.



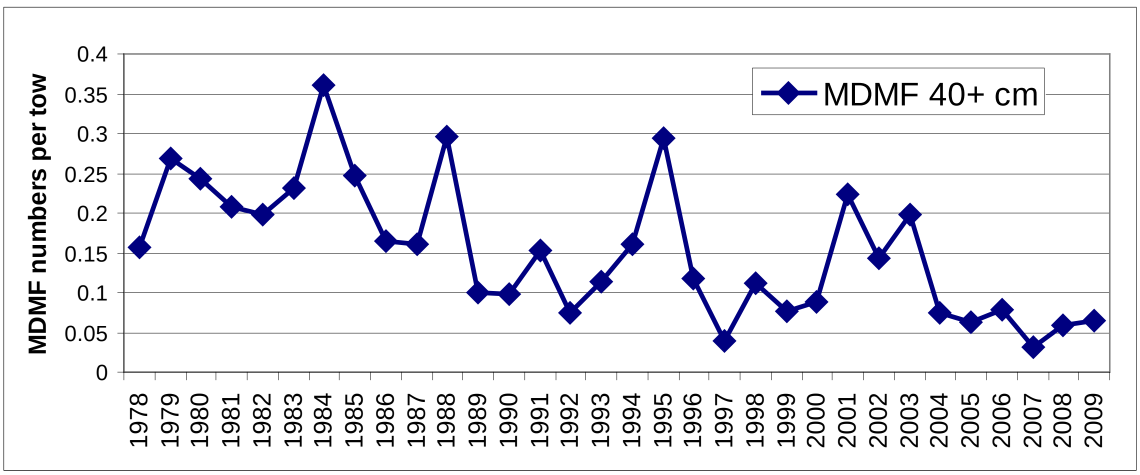
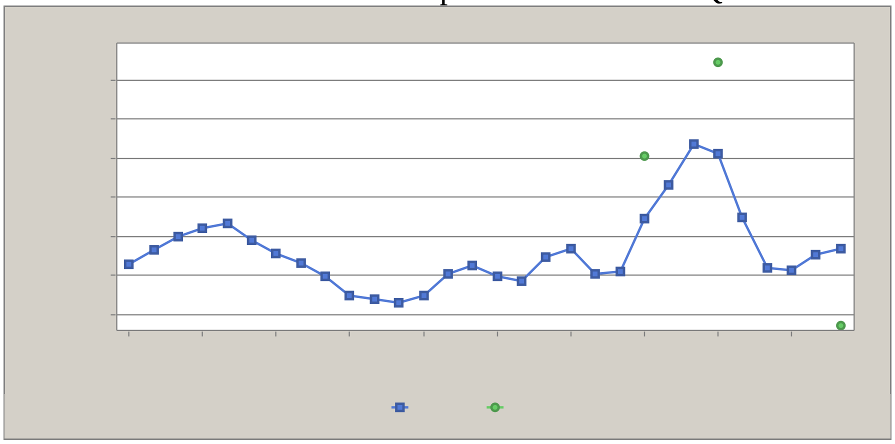
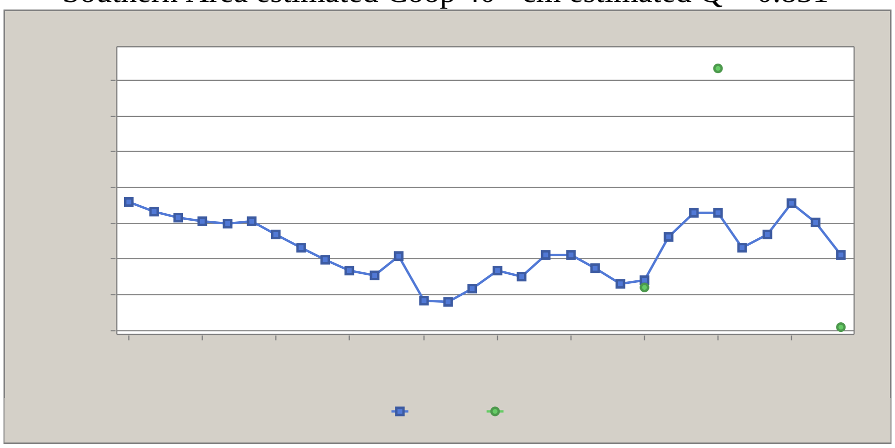


Figure A72. Age 1 (11 to 20cm) and 40+ cm indices from the fall MDMF bottom trawl survey from 1978 to 2009. Many of the years in the age 1 index did not catch any monkfish and relatively low numbers of 40+ cm monkfish are caught per tow.

Northern Area estimated Coop 40+ cm estimated Q = 1.176



Southern Area estimated Coop 40+ cm estimated Q = 0.831



Combined Area estimated Coop 40+ cm estimated Q = 0.679

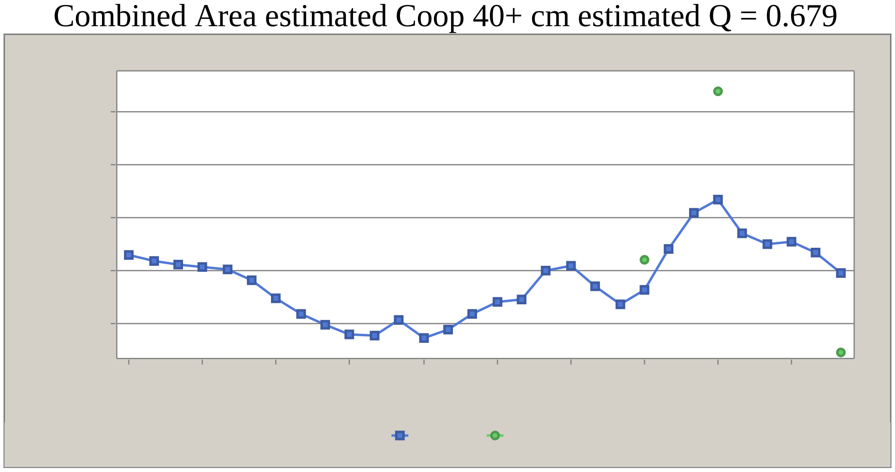


Figure A73. Estimated q’s and fits for the north, south, and combined management area diagnostic runs which incorporated the absolute cooperative 40+ numbers.

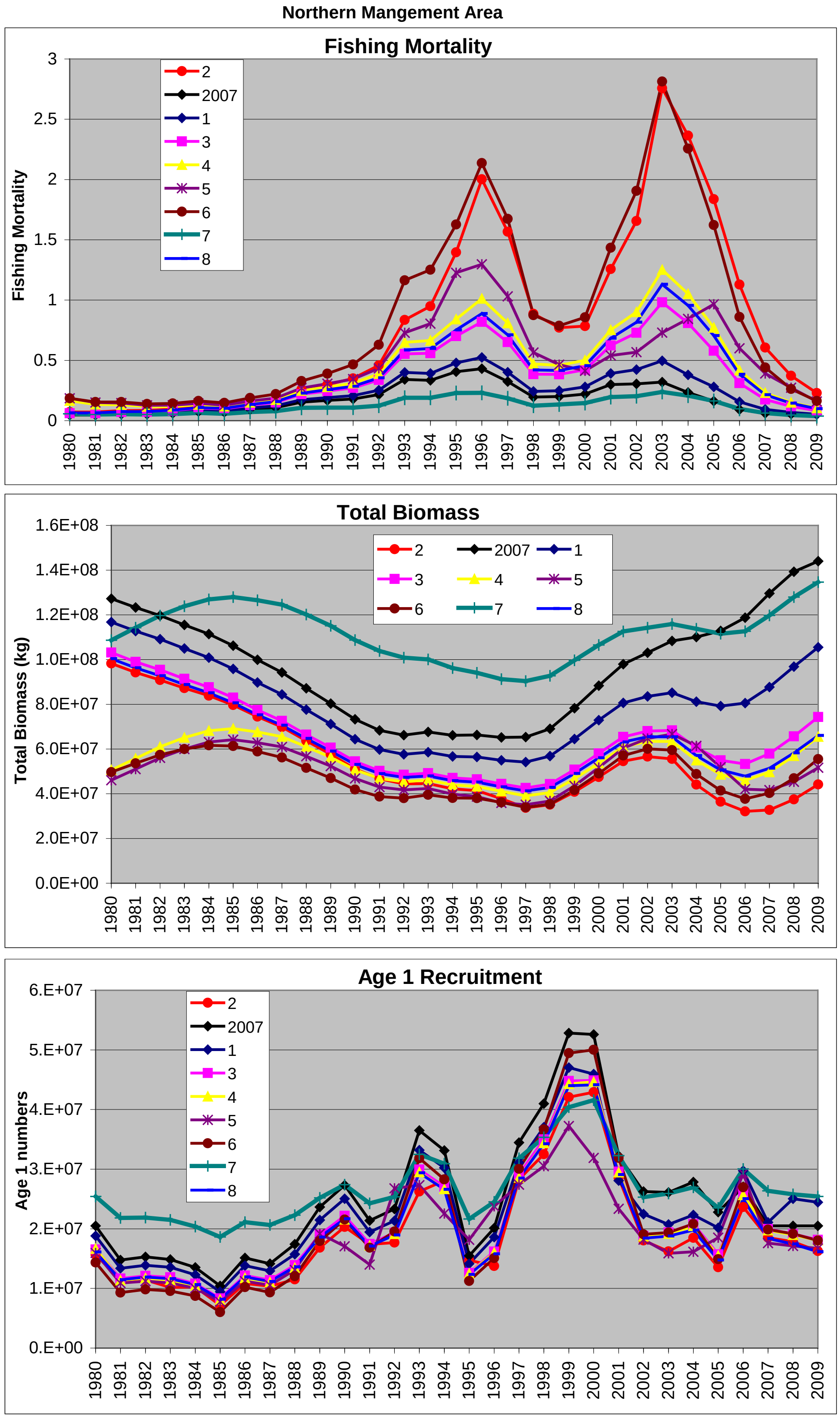


Figure A74. Northern management area monkfish SCALE sensitivity runs (table A3).

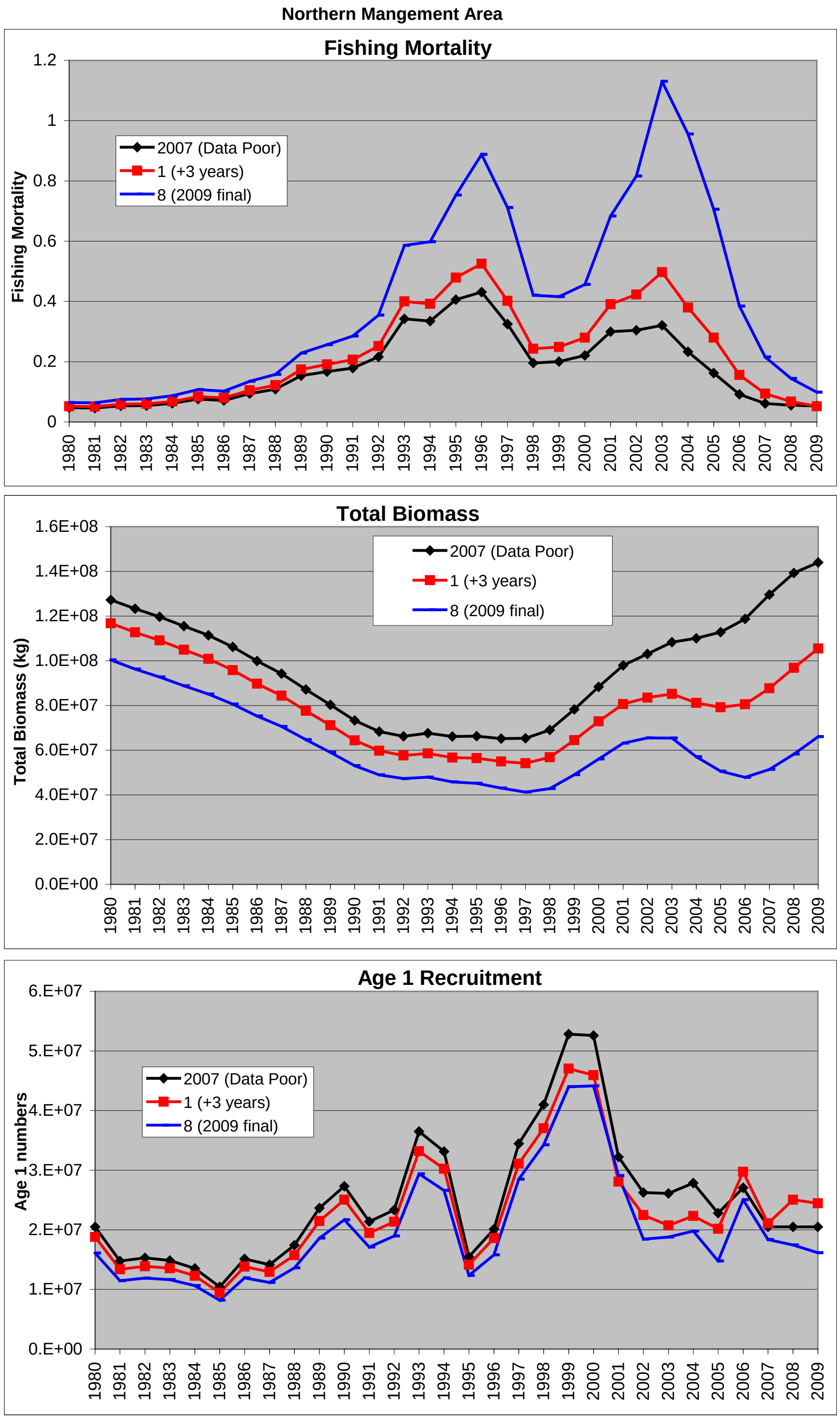


Figure A75. Comparison of northern management area final runs from the 2007 and this assessment. Run 1 (2007 run with updated 2007-2009 data) is also shown.

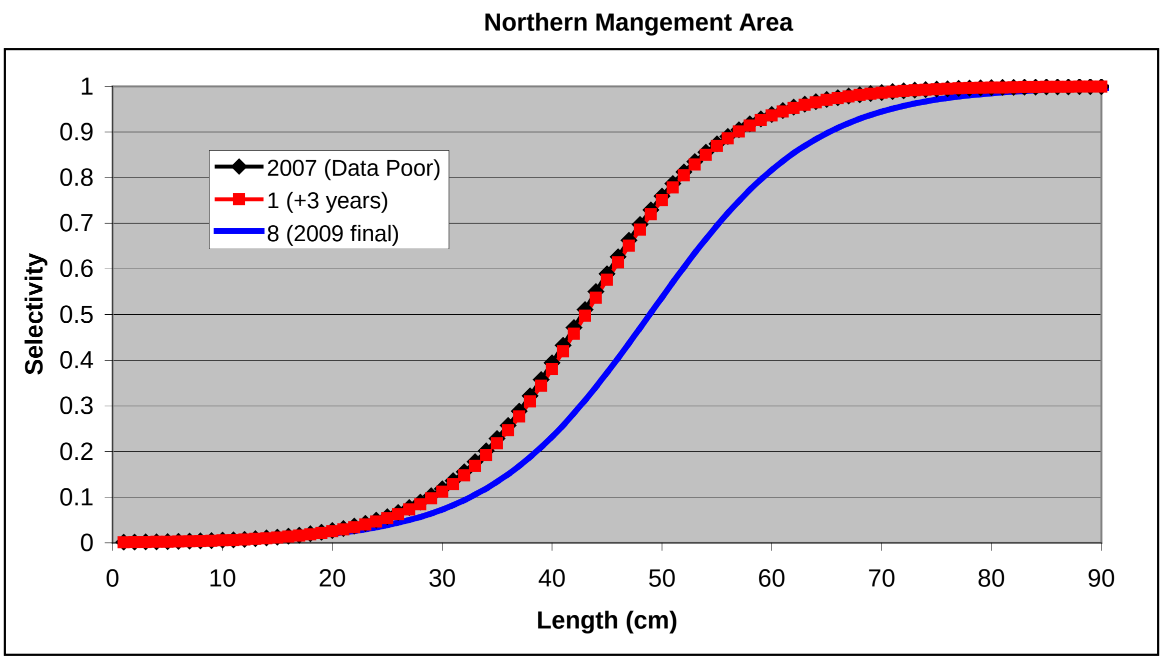


Figure A76. Comparison of northern management area estimated selectivity for the final runs from the 2007 and this assessment. Run 1 (2007 run with updated 2007-2009 data) is also shown.

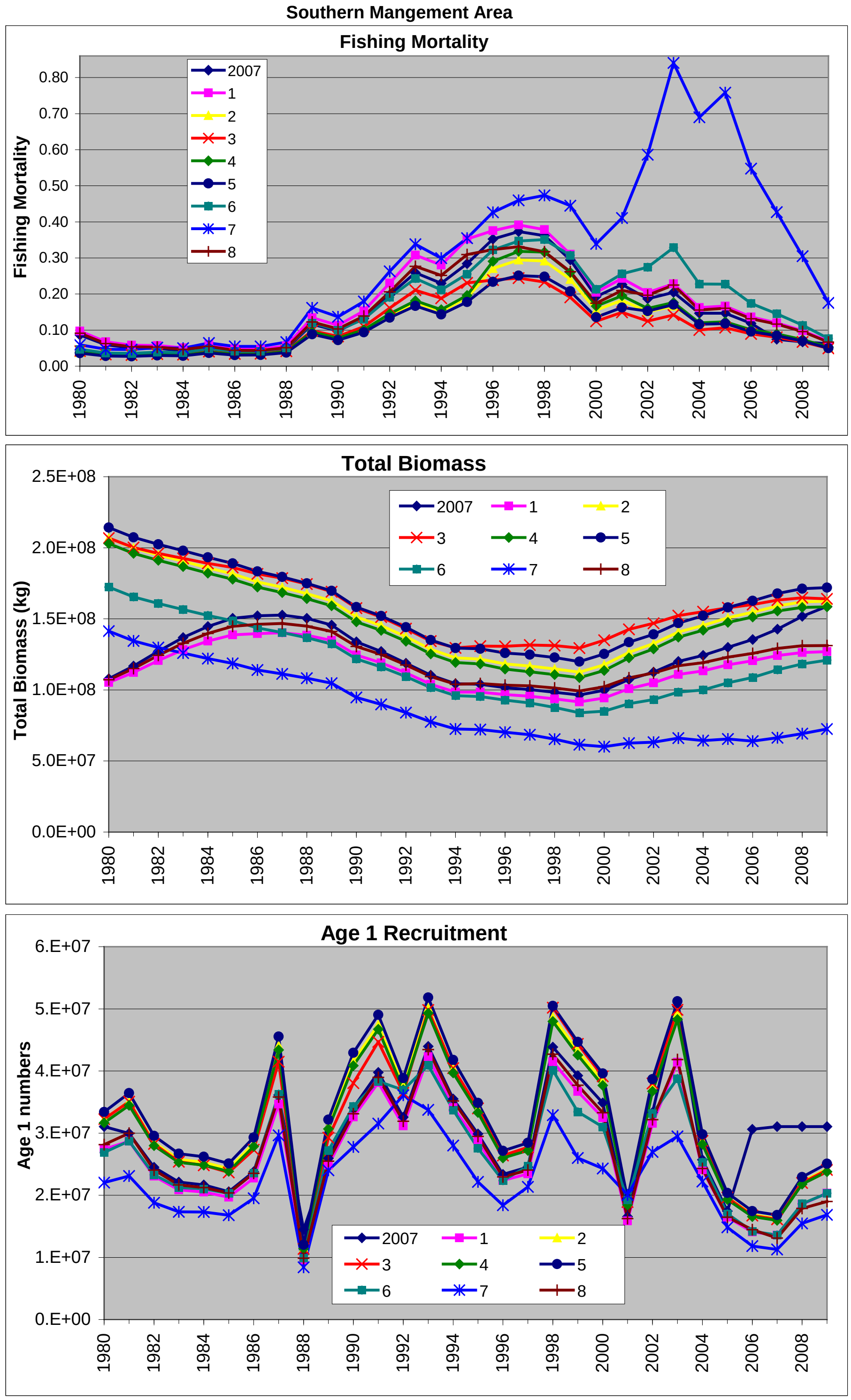


Figure A77. Southern management area monkfish SCALE sensitivity runs.

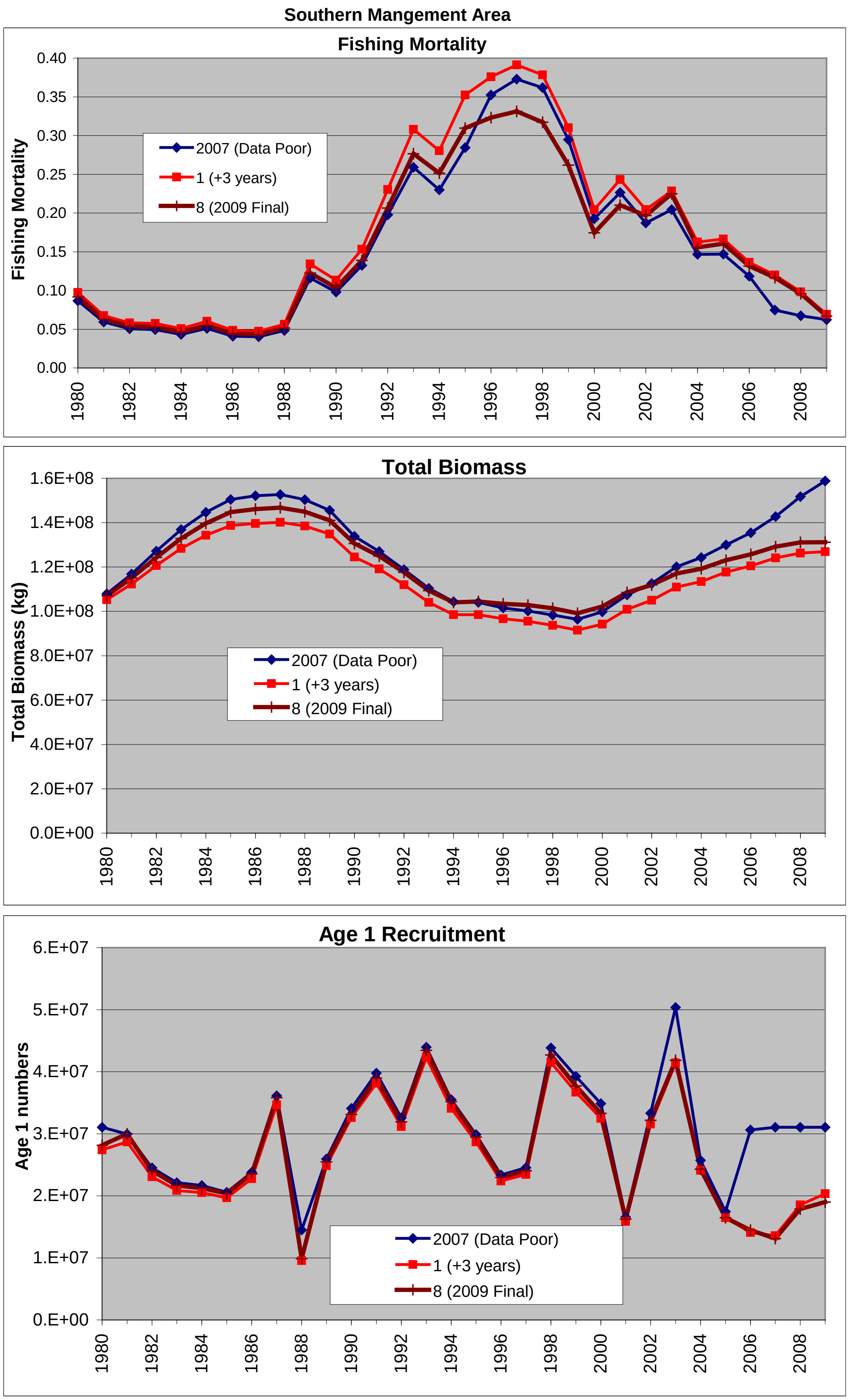


Figure A78. Comparison of southern management area final runs from the 2007 and this assessment. Run 1 (2007 run with updated 2007-2009 data) is also shown.

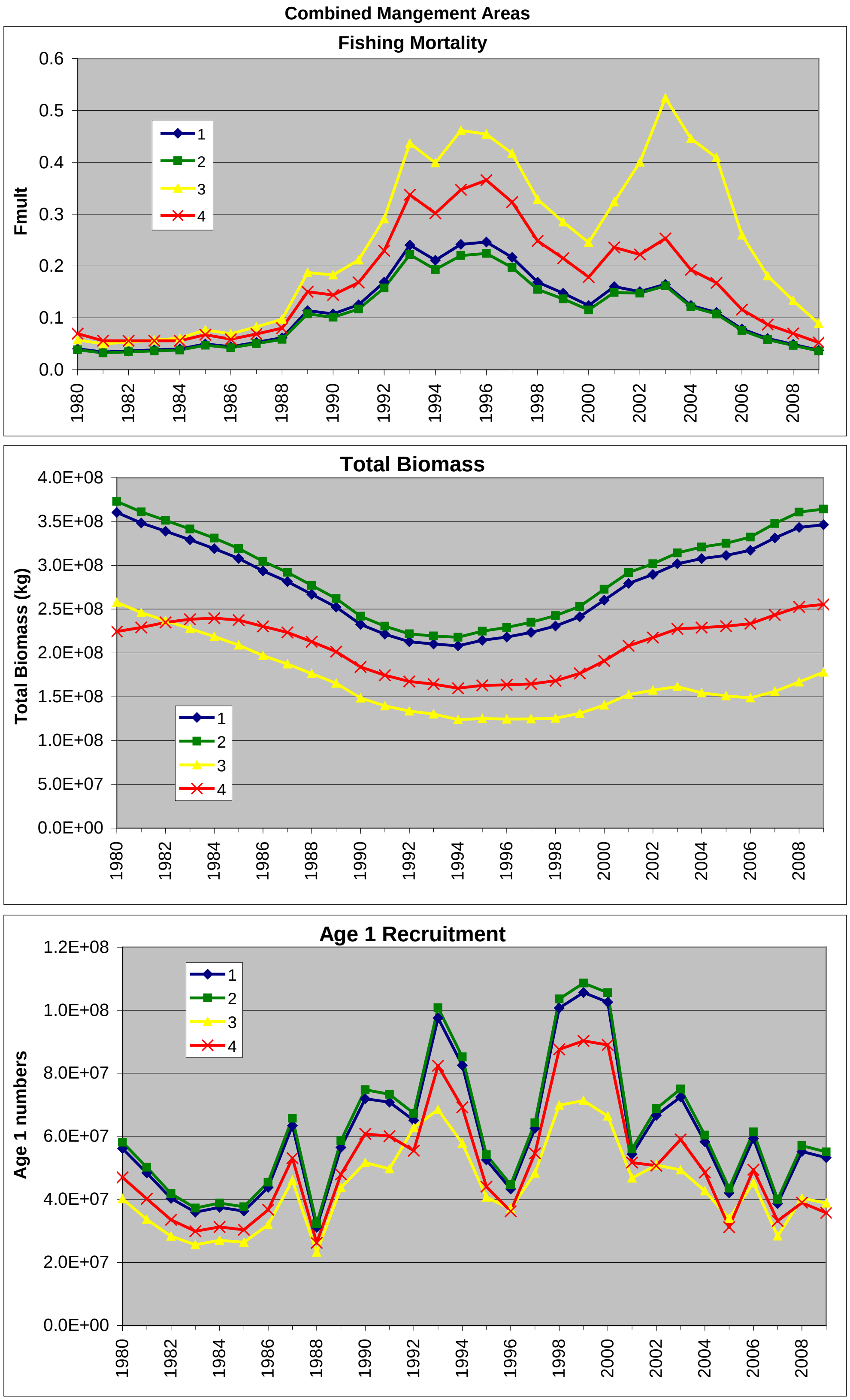
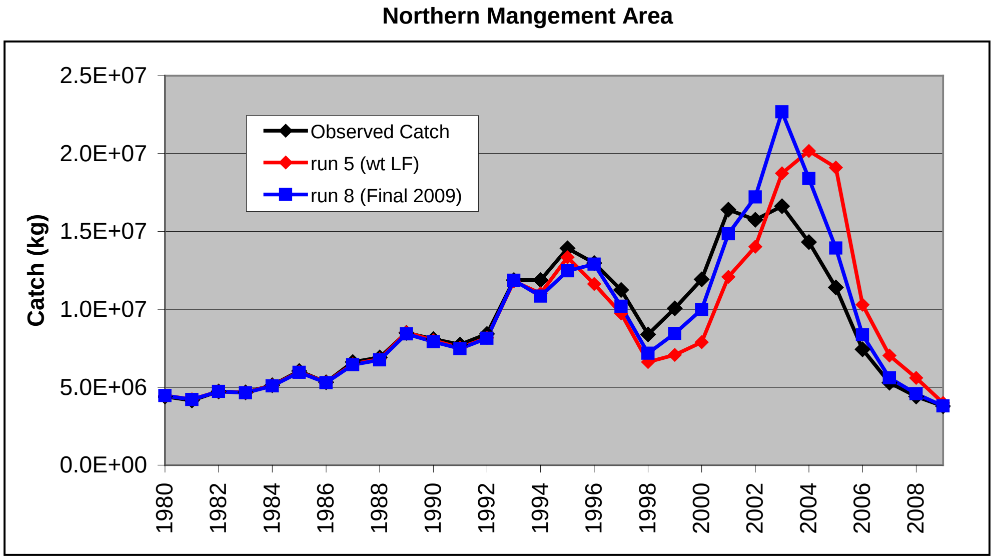


Figure A79. Combined management area monkfish SCALE sensitivity runs.



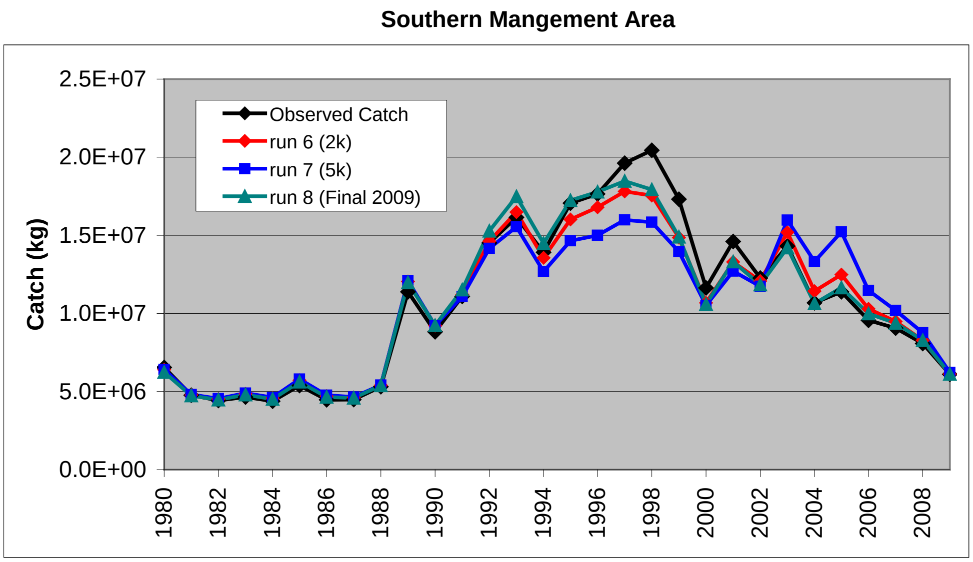


Figure A80. Comparison of northern and southern fits to the catch between the final and sensitivity runs which increased the weighting on fitting the length frequency data.

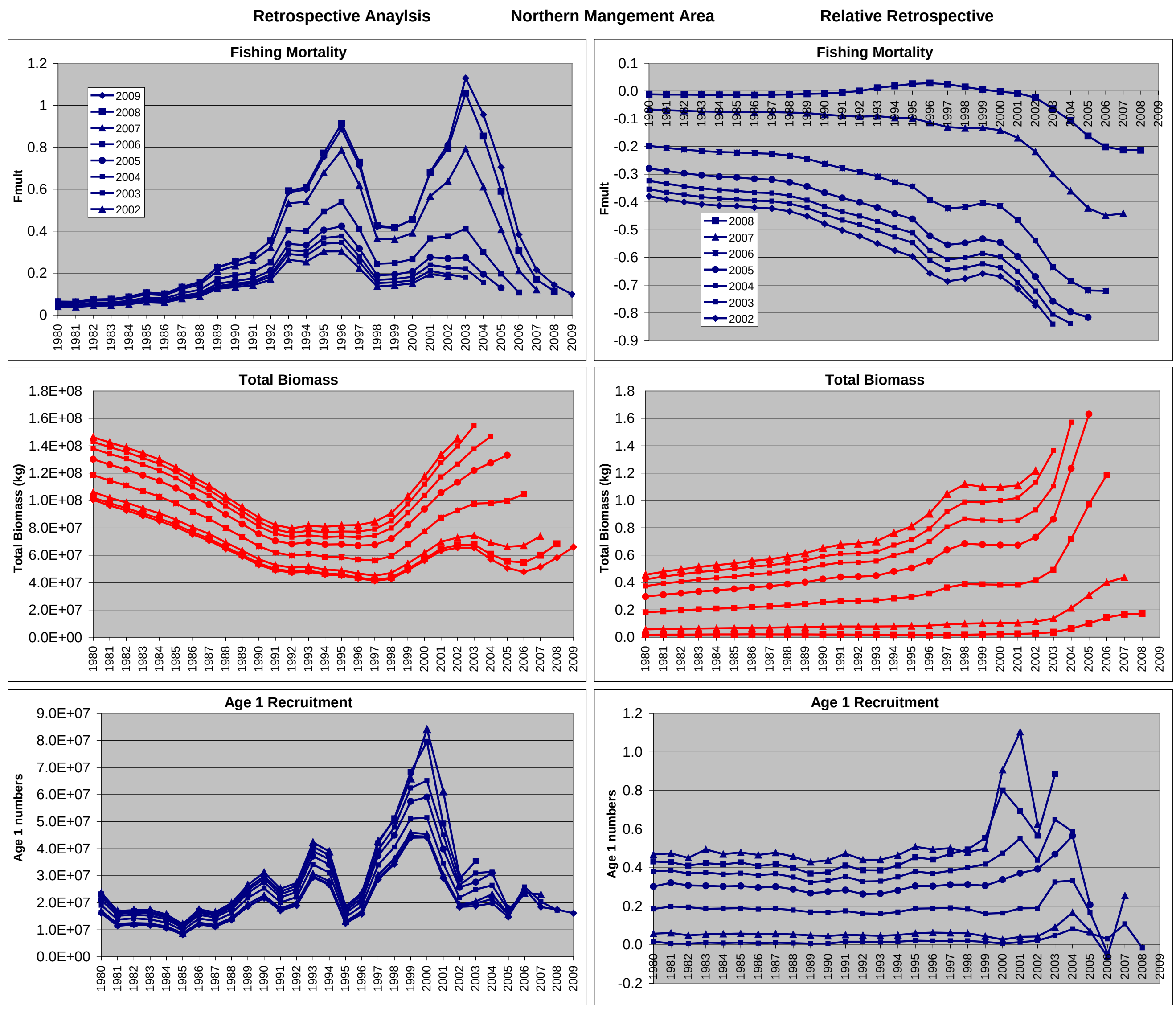


Figure A81. Northern management area retrospective plot for F, total biomass and age 1 recruitment (left). Retrospective relative trends to the terminal year run are on the right.

Retrospective Anaylsis   Southern Mangement Area   Relative Retrospective

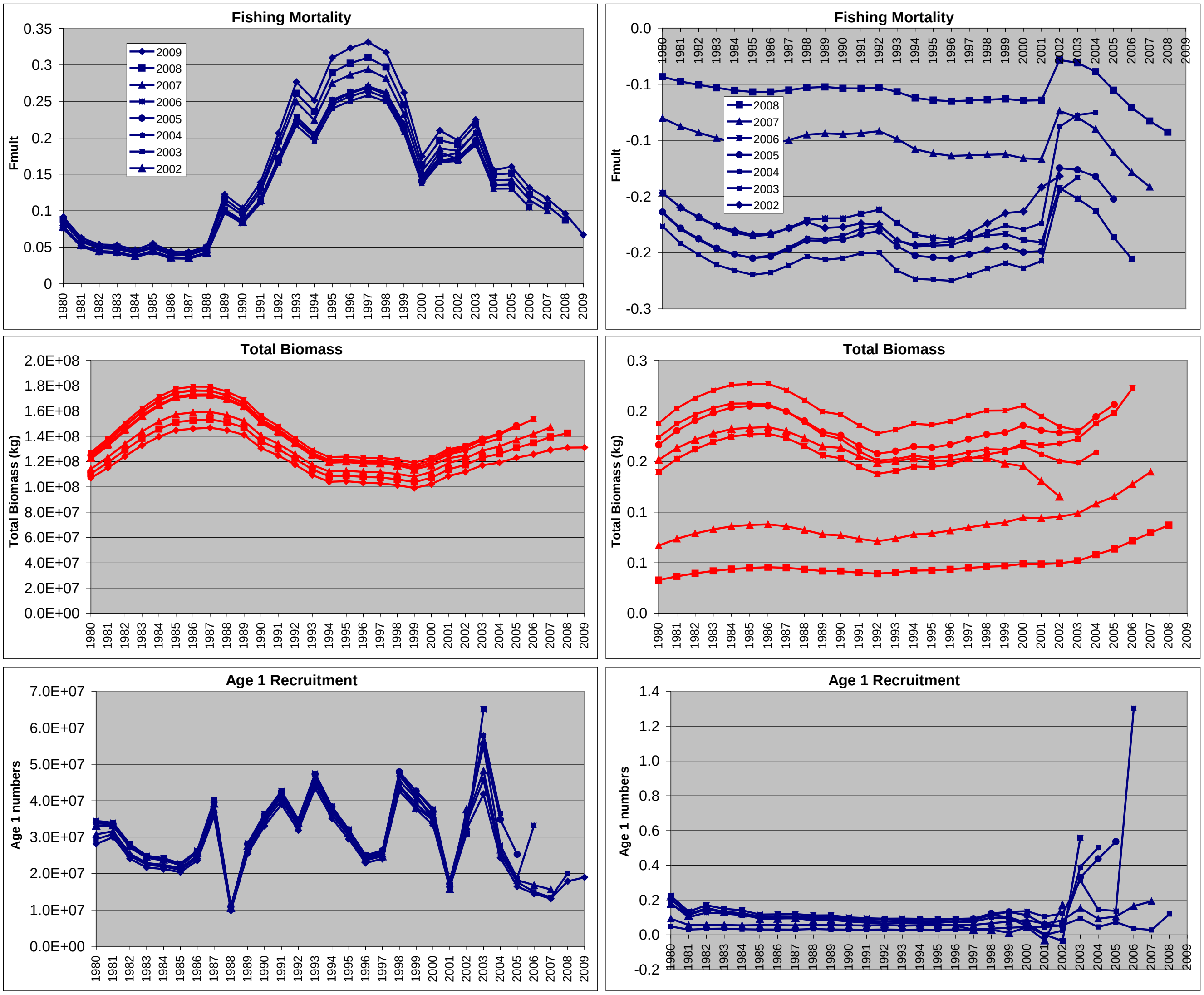


Figure A82. Southern management area retrospective plot for F, total biomass and age 1 recruitment (left). Retrospective relative trends to the terminal year run are on the right.

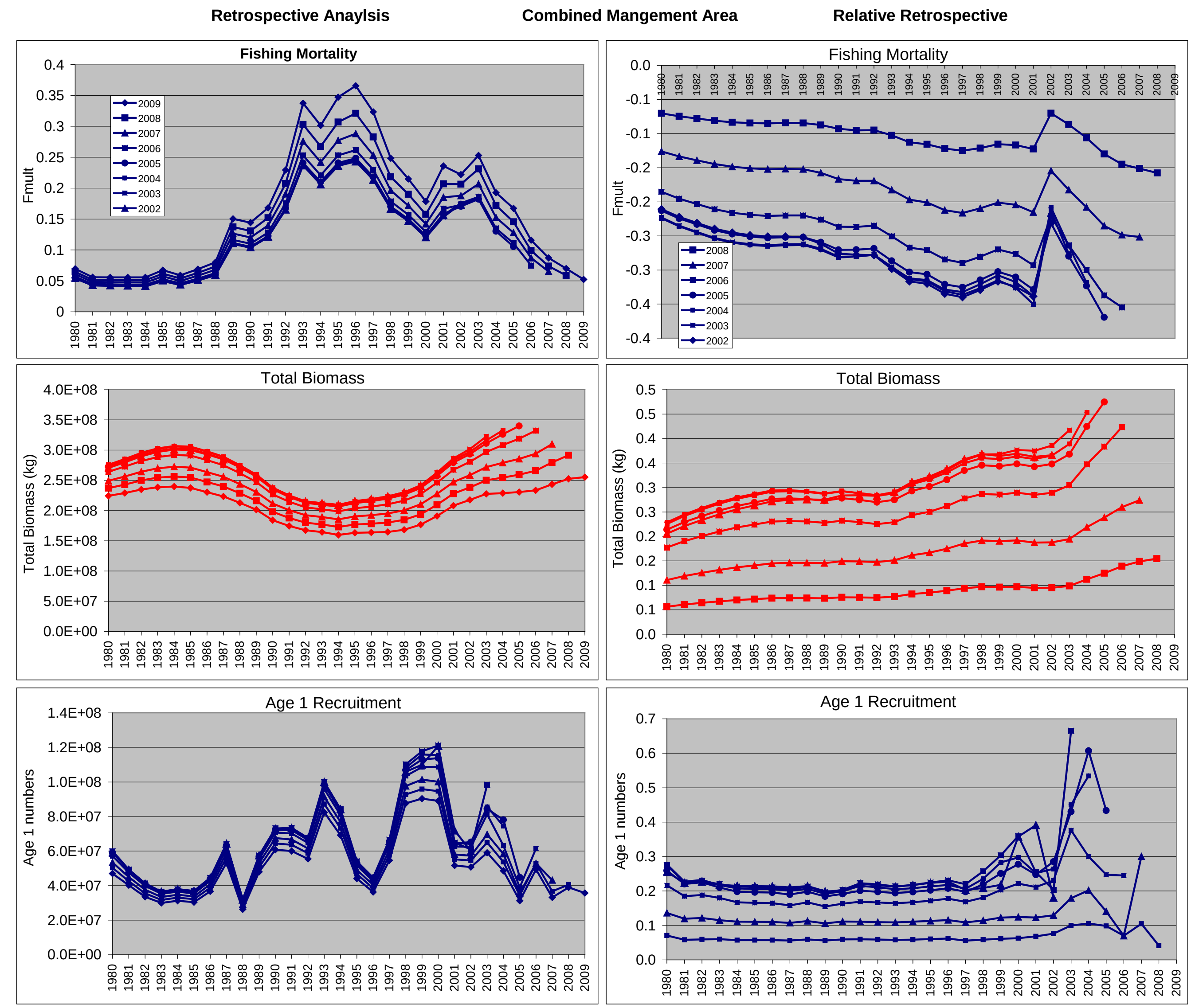


Figure A83. Combined management area retrospective plot for F, total biomass and age 1 recruitment (left). Retrospective relative trends to the terminal year run are on the right.

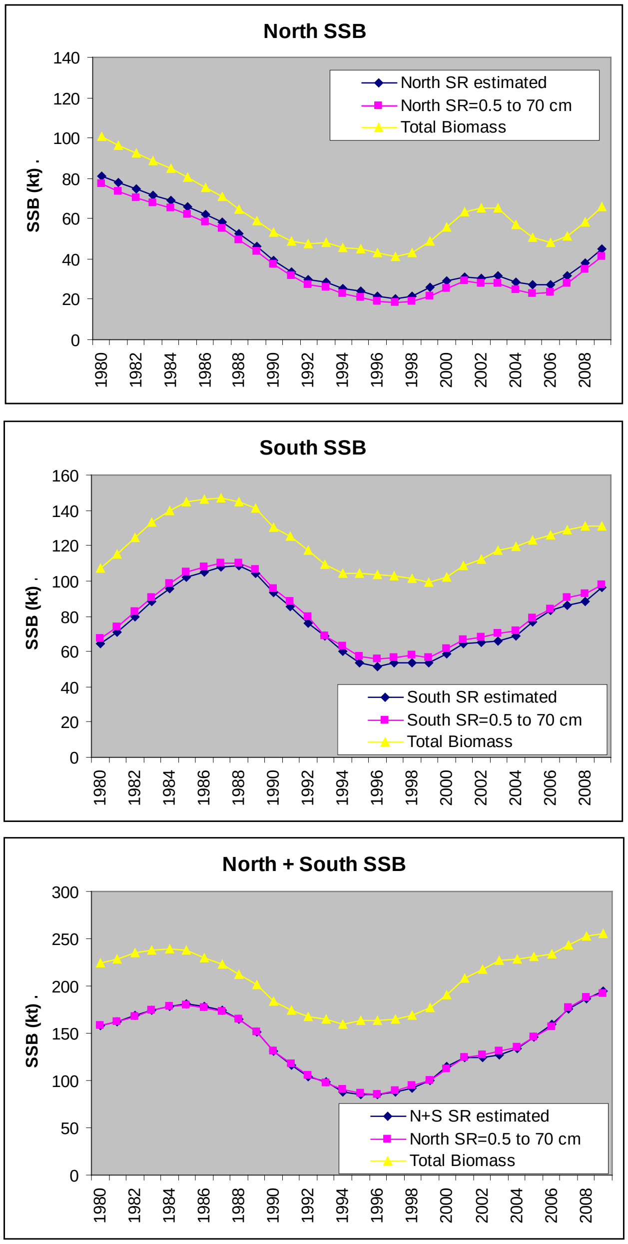


Figure A84. Trends in spawning biomass estimated from SCALE output of numbers at length and applying relationships for maturity at length, weight at length and fraction female at length. Fraction female was estimated from observed ratios at length in survey data (blue diamonds) and assuming 50% female up to 70 cm and 100% female ≥ 70 cm (pink squares).

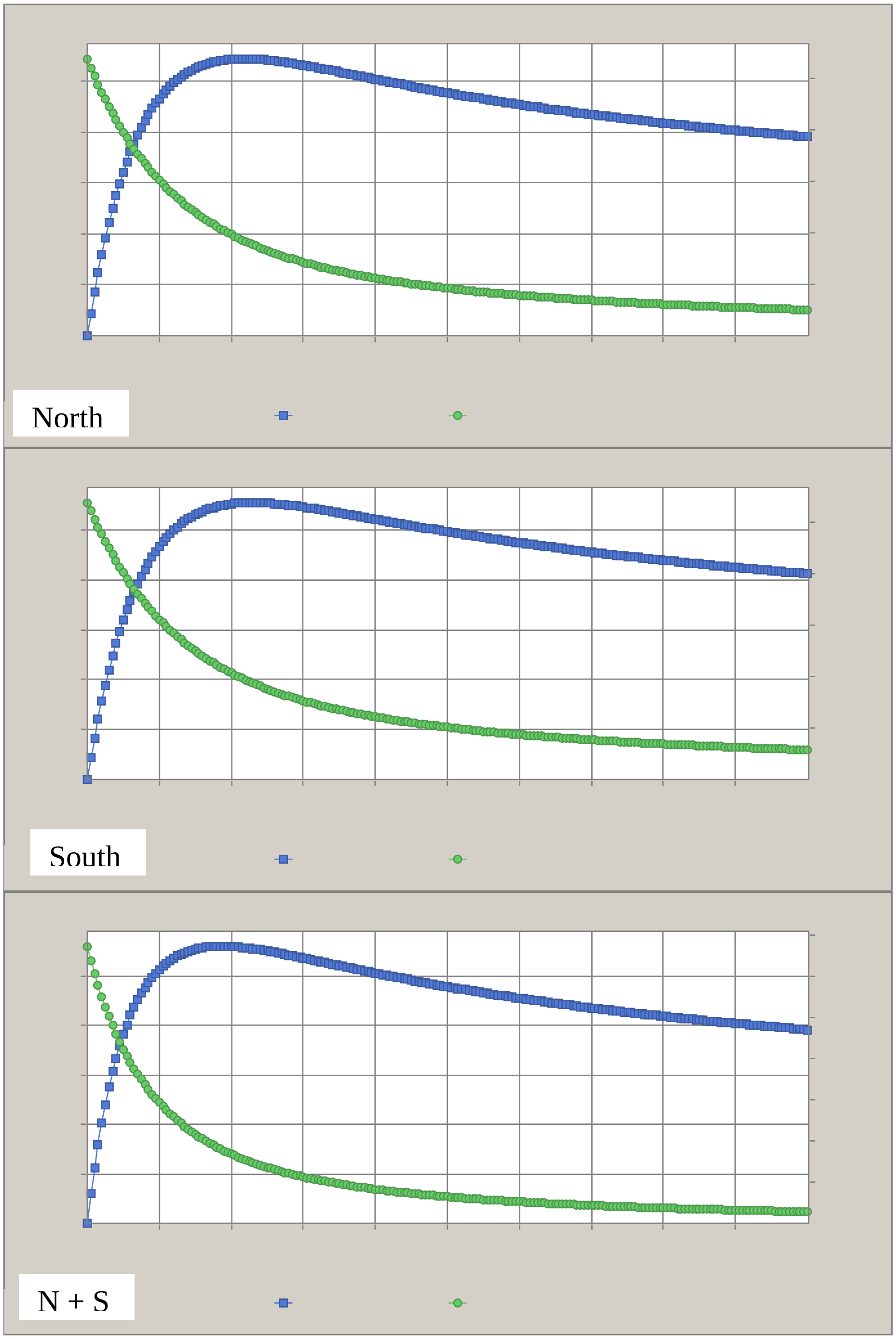


Figure A85. Yield per recruit and spawning stock biomass per recruit curves using selectivity from 2010 SCALE model.

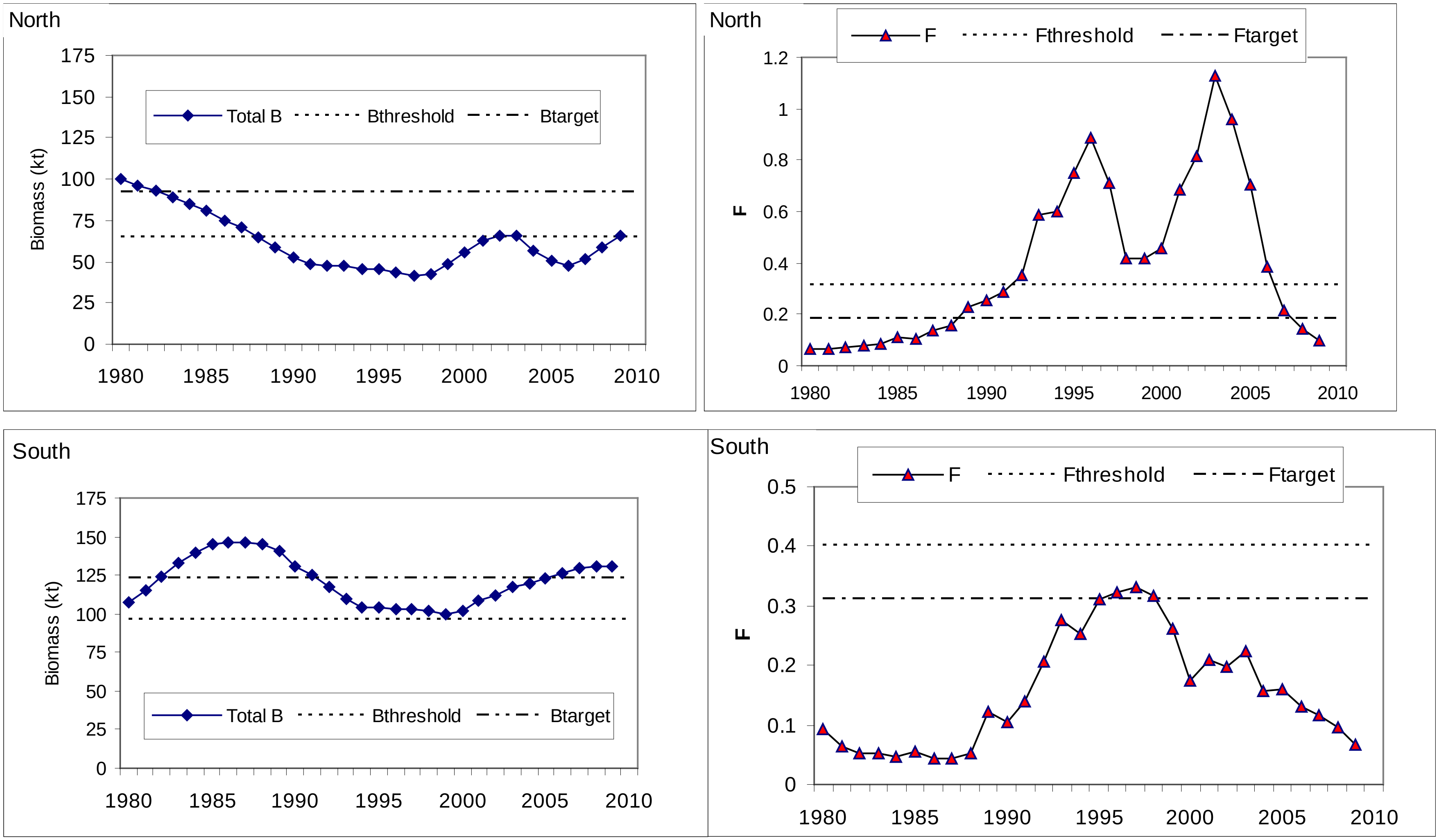


Figure A86. Trends in total biomass and fishing mortality rate (F), from the assessment model (SCALE), relative to the existing (2007) biological reference points for monkfish northern and southern management areas.

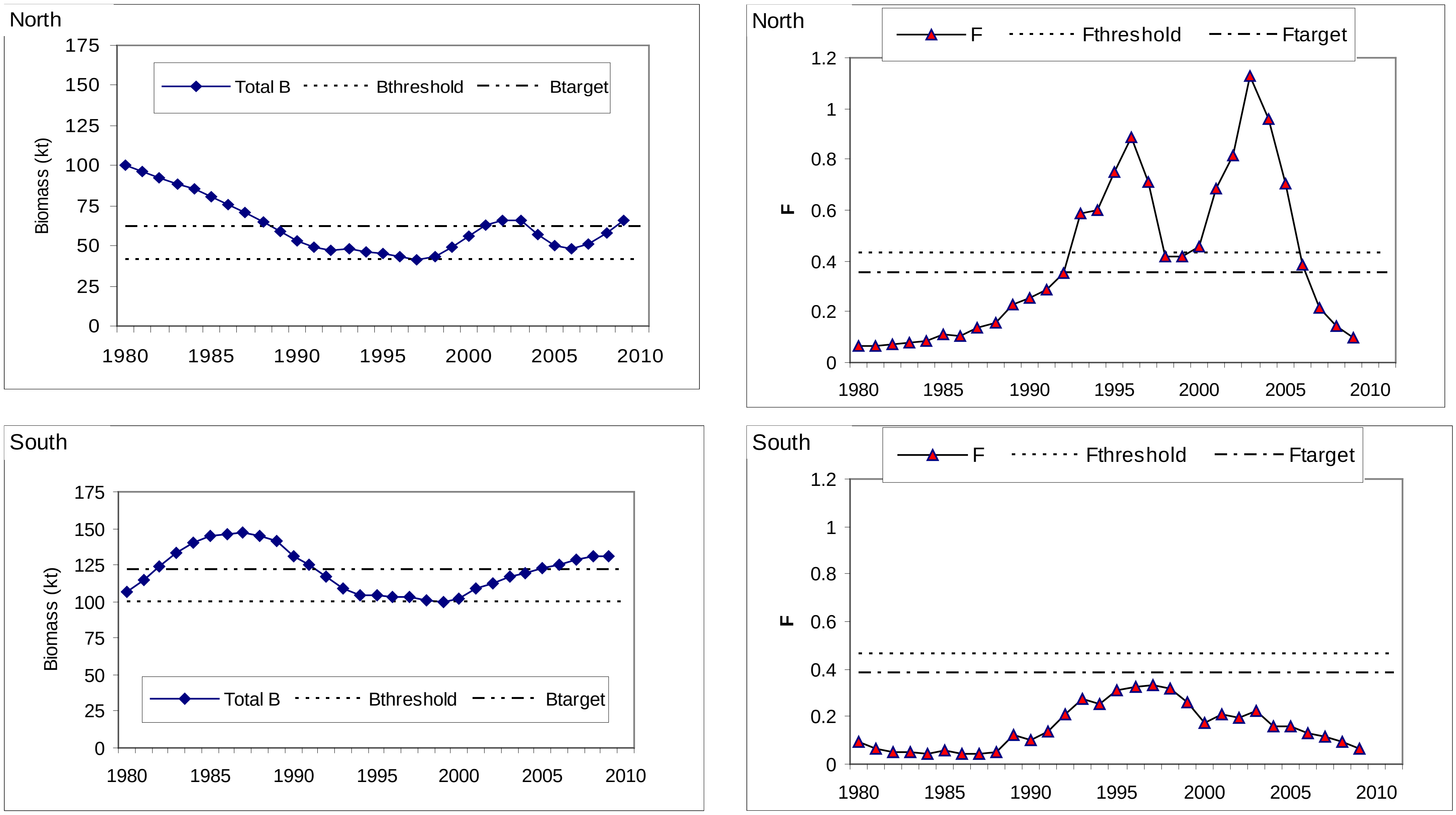
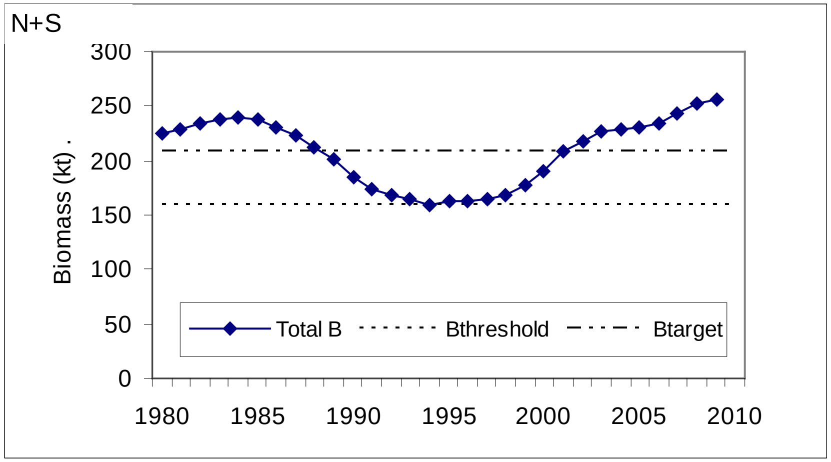


Figure A87. Trends in total biomass and fishing mortality rate (F) from the assessment model (SCALE) relative to updated biological reference points using existing definitions (Bloss, Fmax) for monkfish for northern and southern areas.



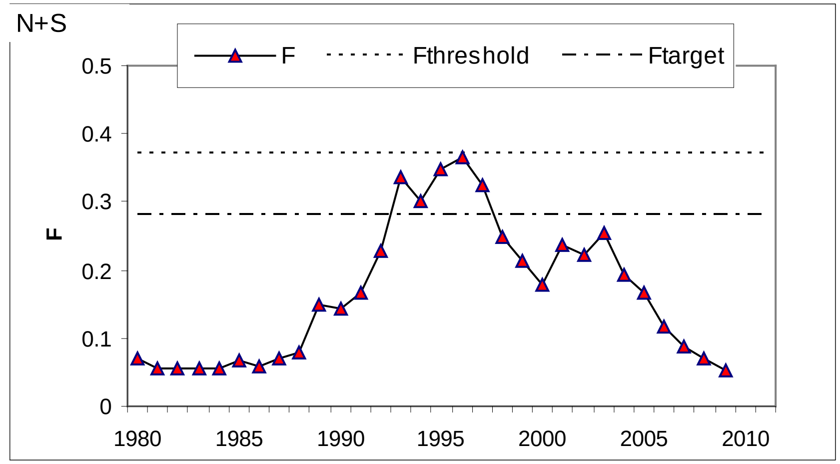


Figure A88. Trends in total biomass and fishing mortality rate (F) from the assessment model (SCALE) relative to updated biological reference points using existing definitions (Bloss, Fmax) for monkfish for combined areas.

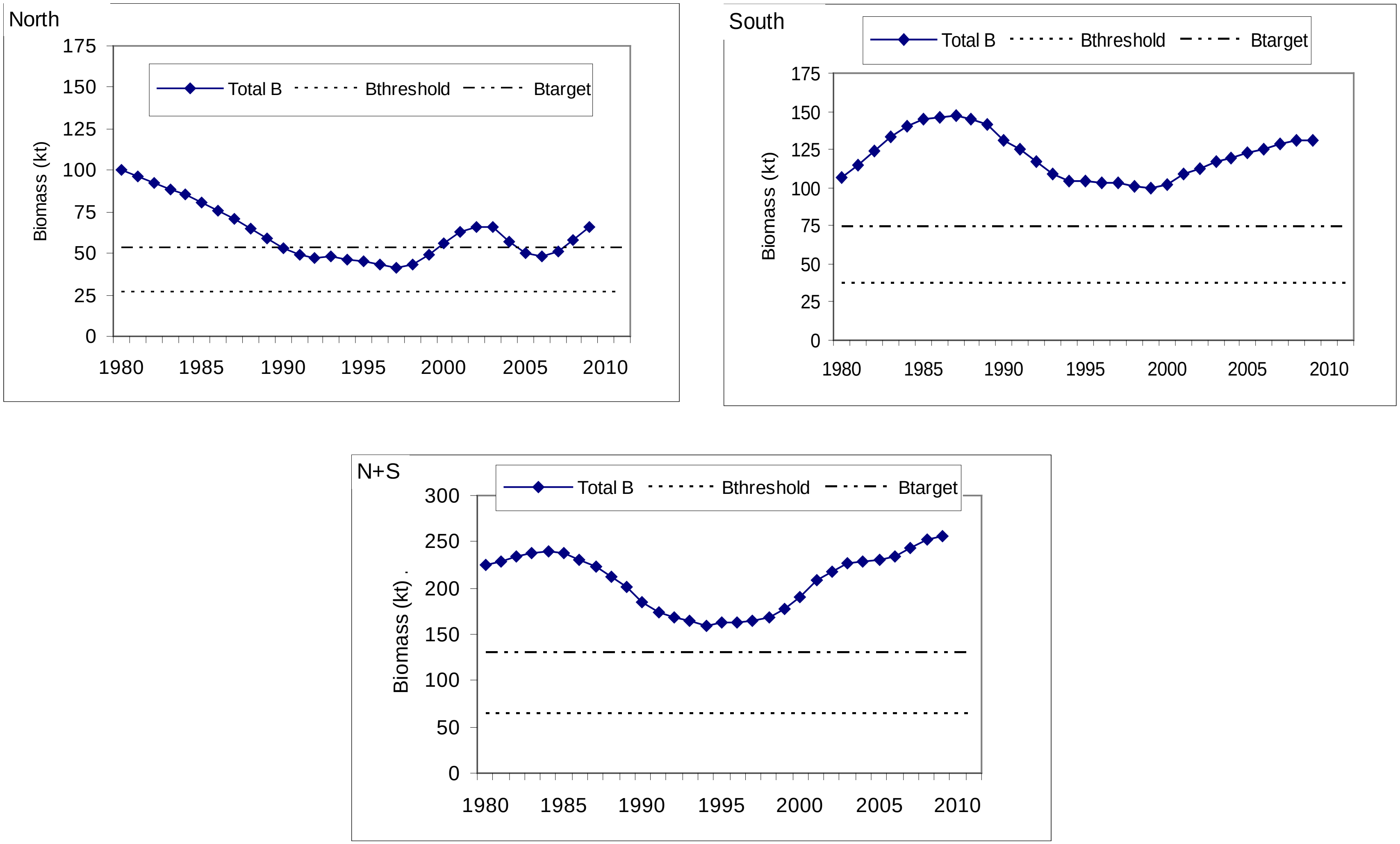


Figure A89. Trends in total biomass from the assessment model (SCALE), relative to the Bmax biological reference points for monkfish for northern, southern and combined management areas.

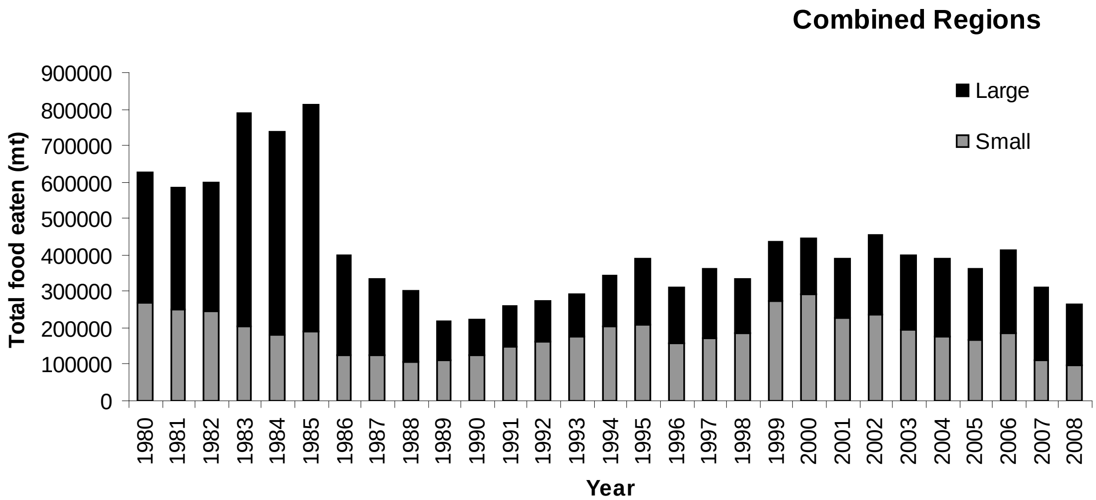


Figure A90. Total amount of food consumed by goosefish.

Initial Analyses of Depletion Experiments for Monkfish

April 12, 2010

Southern Demersal Working Group

SARC 50

Paul Rago

The Catchability Coefficient
A Fundamental Property in Fisheries Science

- Catchability is a scalar that translates indices of relative abundance to absolute abundance.

- The catchability coefficient q was first defined in 1918 by Baranov who called it the "elemental intensity of fishing" which is the fractional reduction in average density per application of a unit of effort.

- Catchability is a parameter in every fisheries model that combines indices of abundance and estimates of total removals. If the model estimates absolute biomass or numbers, then there is a q buried in the equation sets.

- Index = Catchability x Absolute Abundance

- Model-based estimates of catchability can be a source of instability in dynamic models.

Simple Depletion Models
Leslie and Davis 1939, DeLury 1947

Primary assumptions include

1) All extant individuals have the same probability of being caught in a sample,
2) Expected catch in a sample is proportional to sampling effort,
3) Units of sampling effort are independent and additive,
4) Catch depends on the cumulative catch of preceding samples, and
5) All removals are known.

Assumptions are violated in subtle but important ways

- Variations with size of animal
- Changes in availability to gear
- Changes in behavior of animals
- Loss of animals as a result of sampling

The Basic Depletion Model

Catch = P(Capture|Encounter) x P(Encounter) x Population

$$(1) \qquad E(C) = e\left(\frac{a}{A}\right)N$$

Where

e = gear efficiency
a= area swept by unit of gear
A= Total area occupied by population
N = total population size
q = e (a/A)

Note that density D = N/A so that

C = e a D

Deriving the Recursive Model

- Consider a population of size N_o in an area of size A such that the initial density is $D_o = N_o/A$.

Let $C_j = q\, N_{j-1}$ and $N_{j+1} = N_j - C_j$

- Then $C_1 = q\, N_0$ and $N_1 = N_0 - C_1$
- Then $C_2 = q\, N_1$ and $N_2 = N_1 - C_2$
- Then $C_3 = q\, N_2$ and $N_3 = N_2 - C_3$
- Lather, rinse, repeat to get

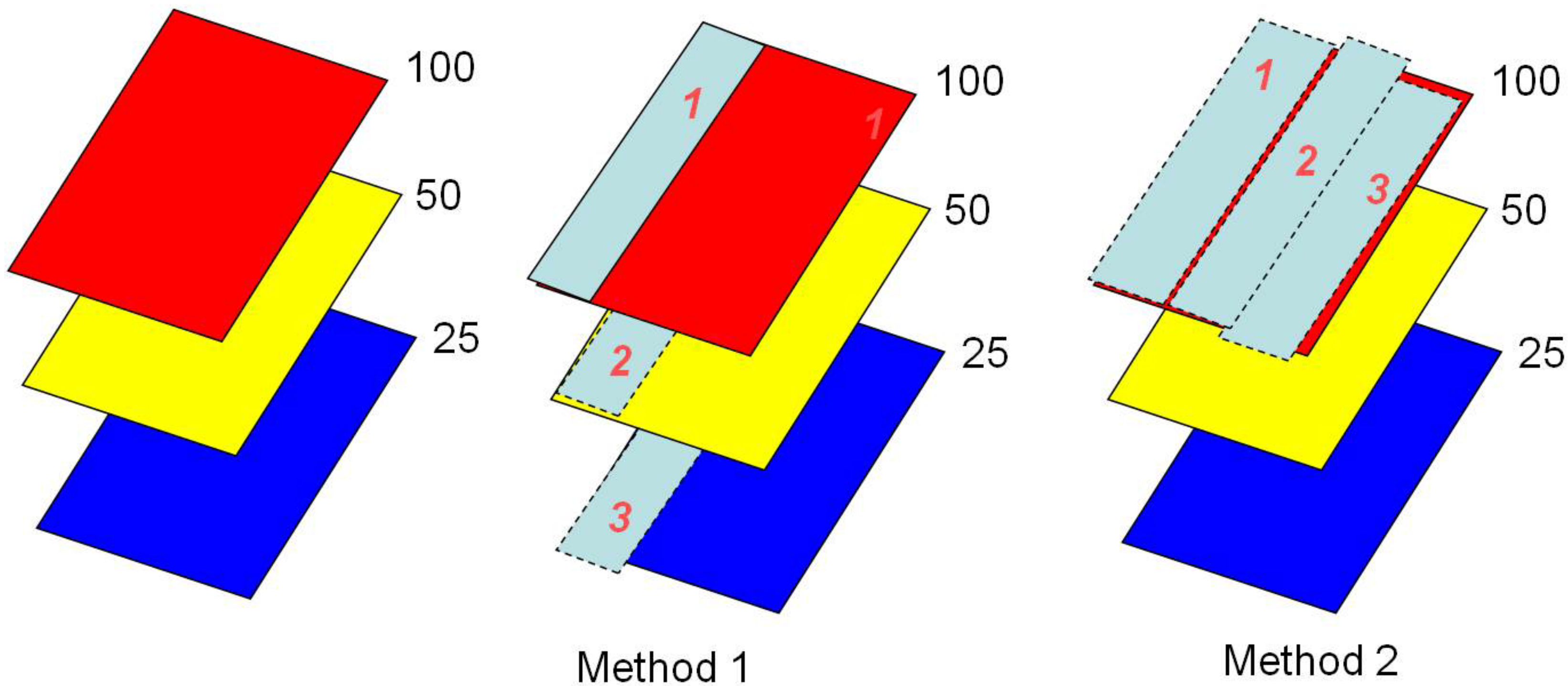
$$E(C_i) = q(N_0 - T_{i-1}) \tag{2}$$

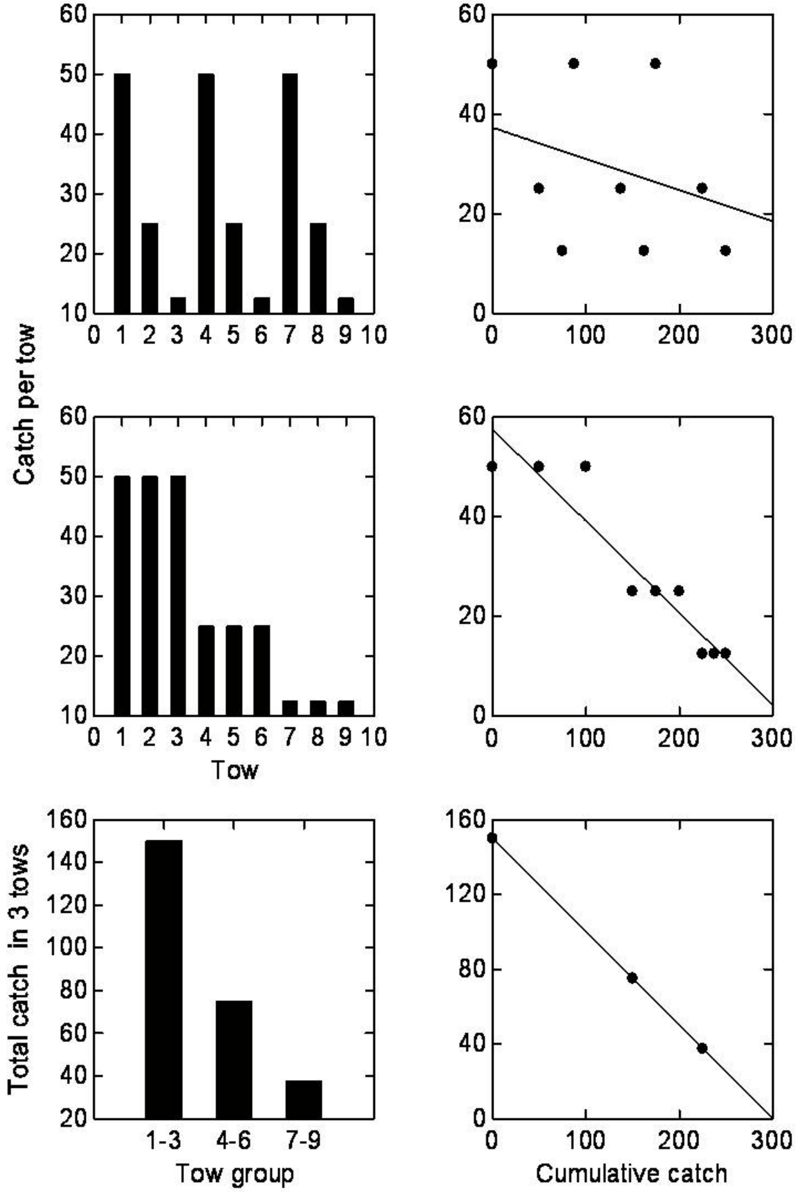
where $T_{i-1} = \sum_{j=1}^{i-1} C_j$,

for $i = \text{tow number}$.

Why Space Matters

- Consider a population that exists over some area A and that the population N is sessile. Assume that the sampling device has a 50% efficiency and that it can sample 1/3 of the area per unit effort.





<u>Toy Example:</u> Why spatial pattern of depletion tows matters.

True Population = 300
True Efficiency=0.5

MLE(1) N=595, Efficiency=0.06

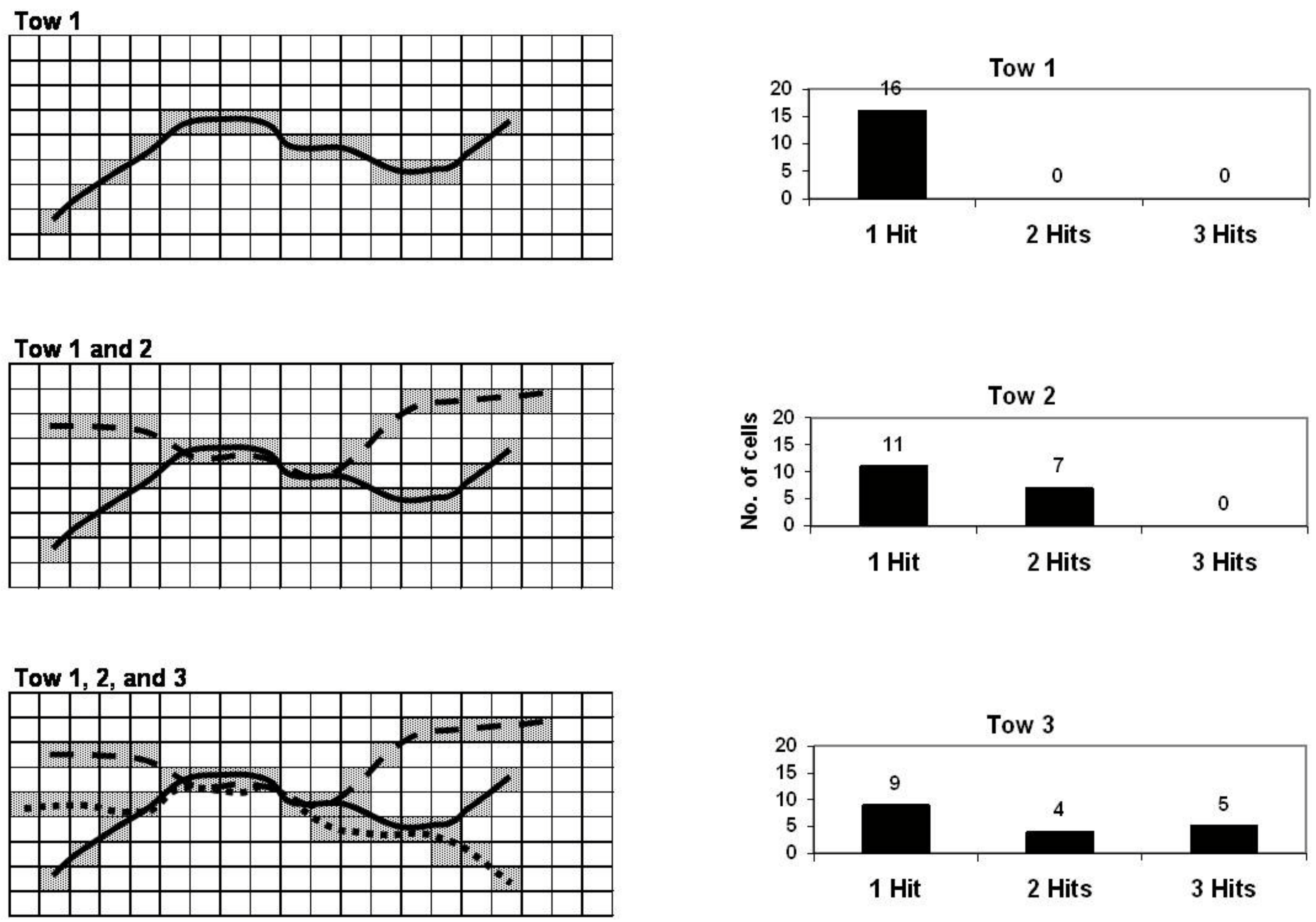
If the catch patterns are incorrectly assumed to be the result of random variability, then MLE estimates are biased.

MLE(2) N=312 Efficiency=0.18

Pooling of samples can work *IF* you know how to aggregate the tows.

MLE(3) N=300 Efficiency=0.50

Consider a spatial pattern where the precision of sampling cannot be controlled but it is possible to “know” where the gear is, after the sample is taken. Clearly, the expected catch is a function of the amount of overlap between successive tows.



To solve the problem, it is necessary to recast the depletion problem in terms of density per unit area, AND to consider a sample as a string of quadrats or patches.

N_0
$N_1 = N_0 - C_1$ $= N_0 - e\,(a/A)N_0$ $= N_0\,(1 - e\,(a/A))$
$N_2 = N_1 - C_2$ $= N_0(1-e(a/A)) - e\,(a/A)N_0\,(1-e(a/A))$ $= N_0\,(1-e(a/A))\,(1-e(a/A))$ $= N_0\,(1-e(a/A))^2$
$N_3 = N_2 - C_3$ $= N_0\,(1-e(a/A))^2 - e\,(a/A)N_0\,(1-e\,(a/A))^2$ $= N_0\,(1-e(a/A))^2\,(1-e(a/A))$ $= N_0\,(1-e(a/A))^3$

$D_0 = N_0/A$
$D_1 = N_1/A$ $= (N_0/A)\,(1-e\,(a/A))$
$D_2 = N_2/A$ $= (N_0/A)\,(1-e\,(a/A))^2$ $= D_0\,(1-e\,(a/A))^2$
$D_3 = N_3/A$ $= (N_0/A)\,(1-e\,(a/A))^3$ $= D_0\,(1-e\,(a/A))^3$

The expected catch as a function of patch density.

- Via the miracle of recursive pluggation

(3) $$E(C_j) = e\,a\,D_o\left(1 - e\left(\frac{a}{A}\right)\right)^{j-1}$$

- Now we need to extend this concept to a tow where a tow consists of a set of contiguous patches linked together. Record the number of times that a tow "hits" a particular patch. The first tow will consist of a set of quadrats with one hit each. The second tow, if it exactly overlaps the first, will consist of a set of quadrats with two hits each. Otherwise it will have a mix of 1 and 2 hit cells. The third tow can have some mix of 1, 2 or 3 hits. Etc. With some algebra you get

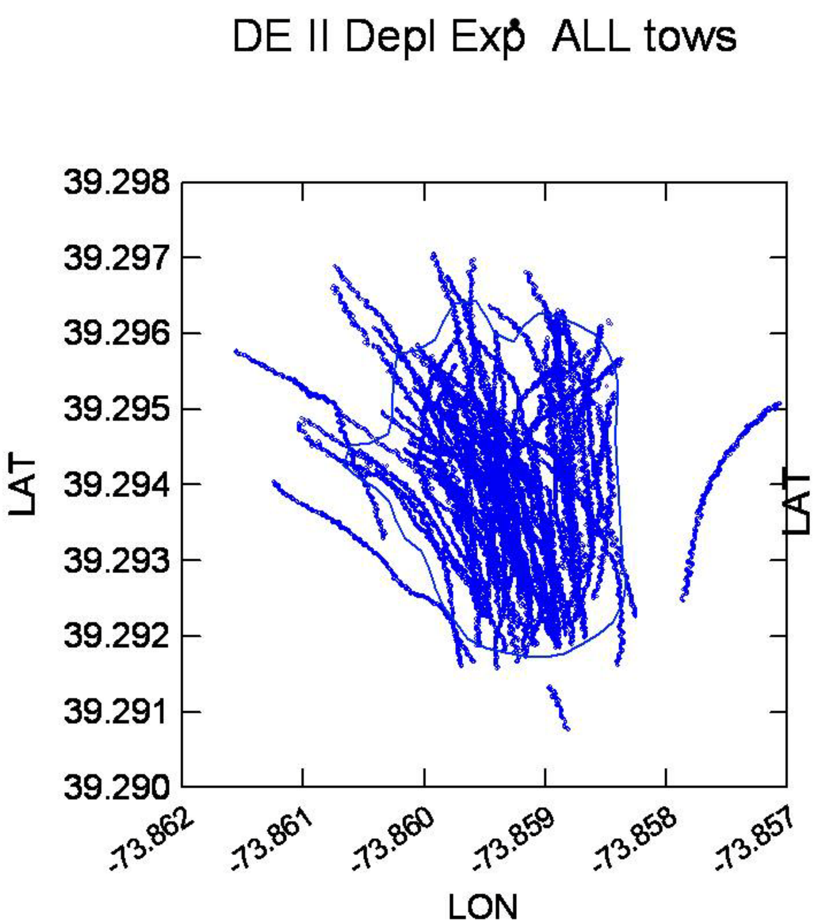
(4) $$E(C_i) = e\,a_i\,D_o \sum_{j=1}^{i} f_{i,j}\,(1 - e\gamma)^{j-1}$$

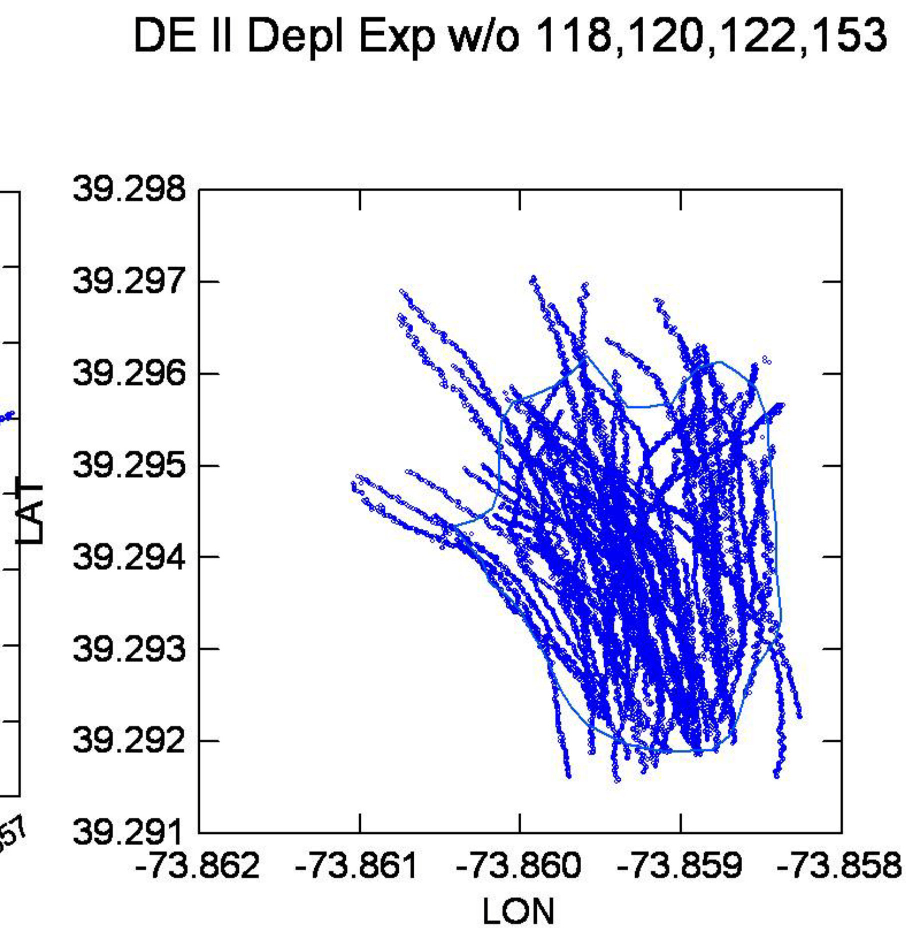
OK, where did the gamma come from and what is $f_{i,j}$?

(4) $$E(C_i) = e\,a_i\,D_o \sum_{j=1}^{i} f_{i,j}(1 - e\gamma)^{j-1}$$

- $f_{i,j}$ is the fraction of a tow of swept area a_i that is hit j times
- Gamma is ratio of the sampling device width over the width of the patch. If the width of sampler is ½ the width of the cell, then gamma =0.5. Thus the patch size can increase in response to increasing uncertainty in the actual position of the sampling gear.
- Gamma can also be used to account for indirect effects of sampling which may occur in a variety of ways. More on this later.

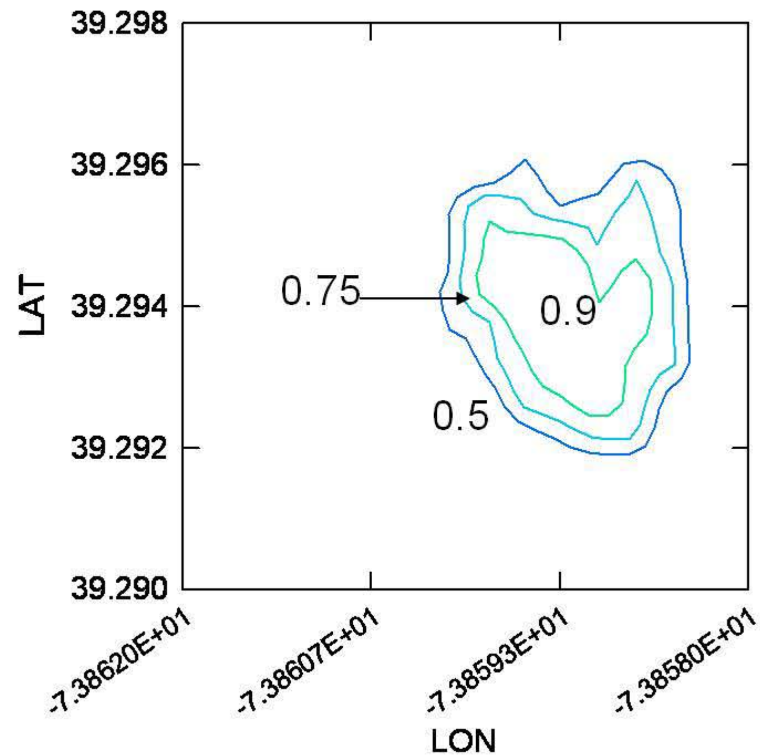
Deriving the expected catch for a set of randomly overlapping tows. A depletion experiment on DE II

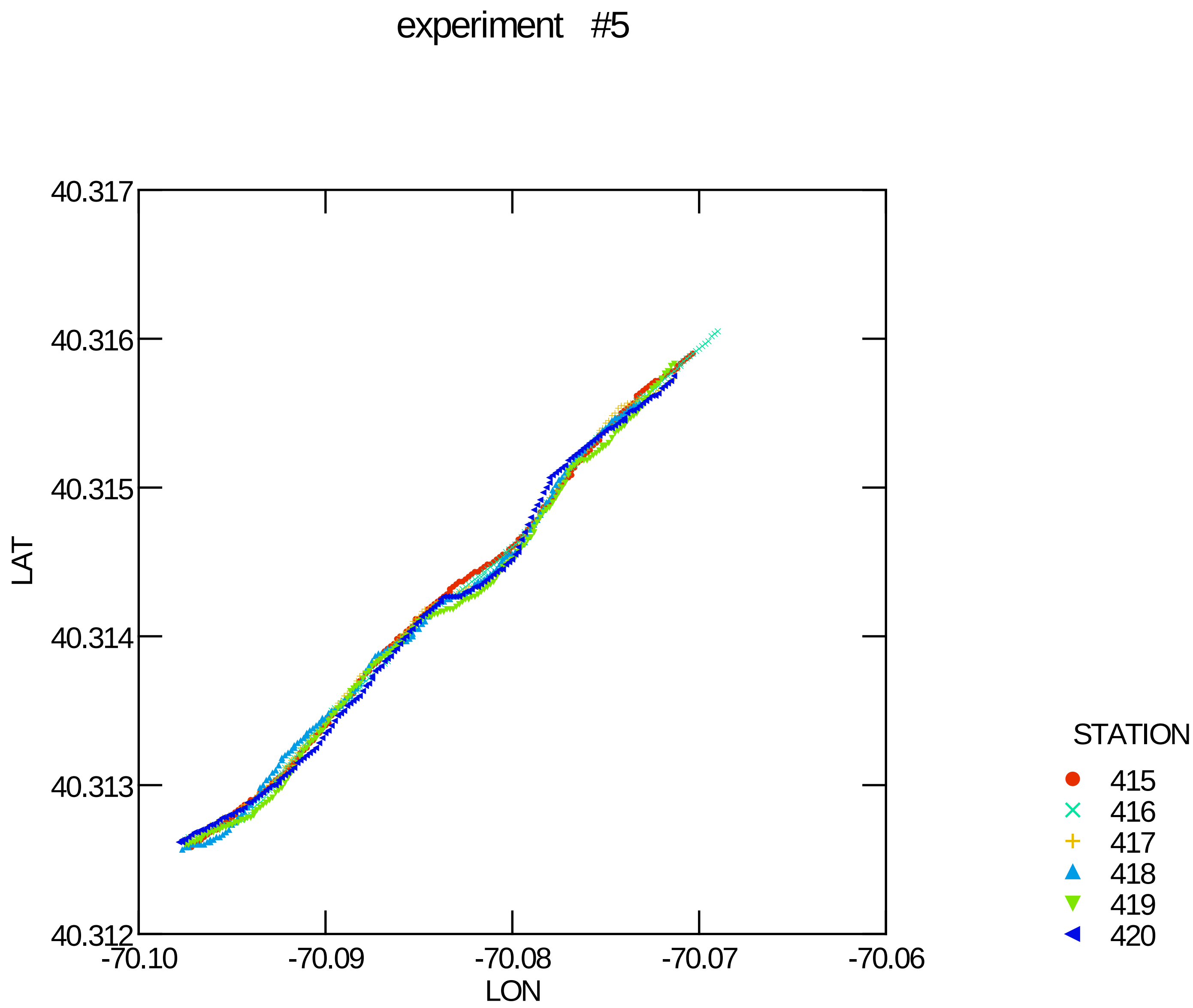


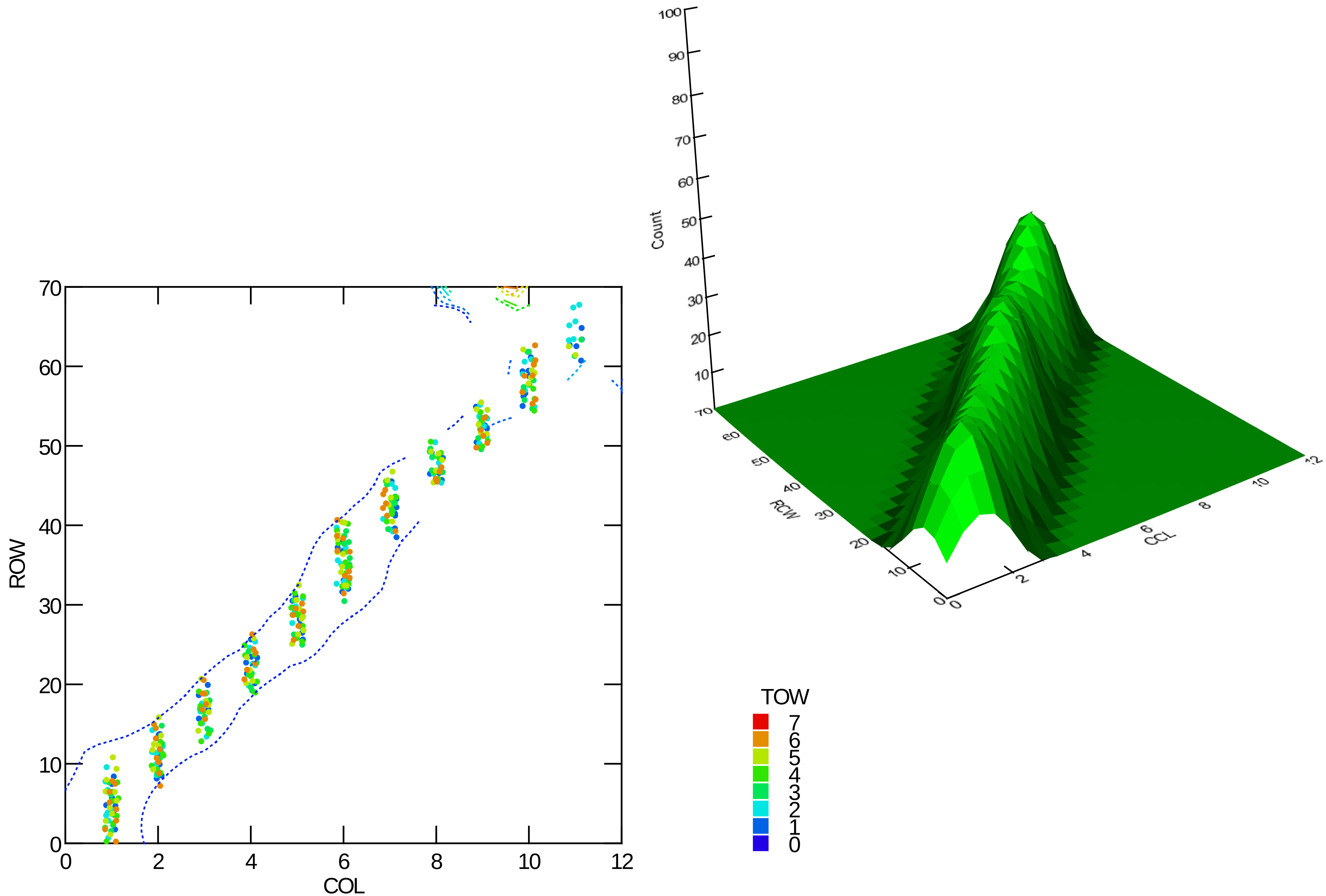


A kernel density can be used to illustrate to concentration of sampling intensity.

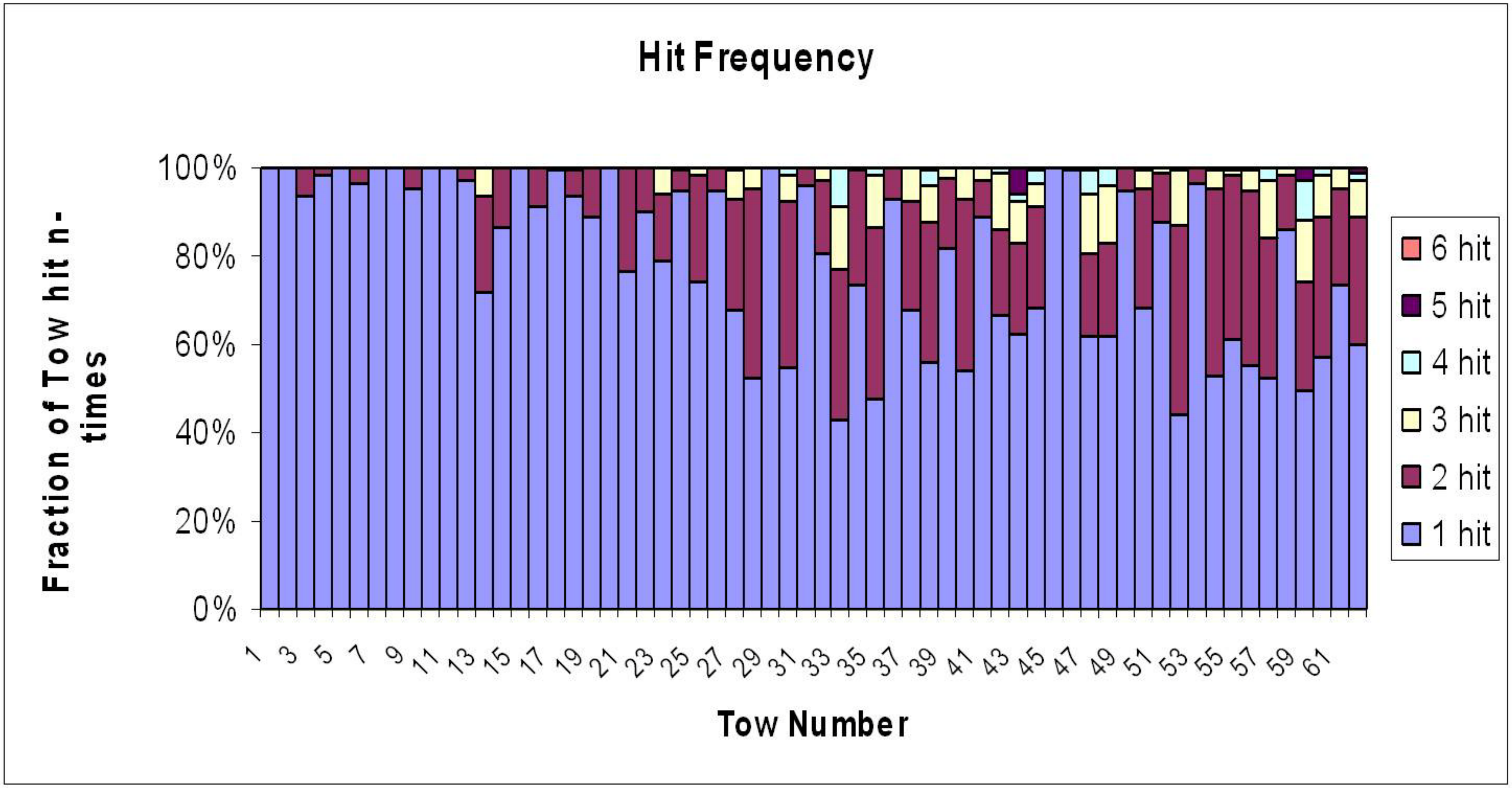
- DE II Depl Exp w/o 118,120,122,153



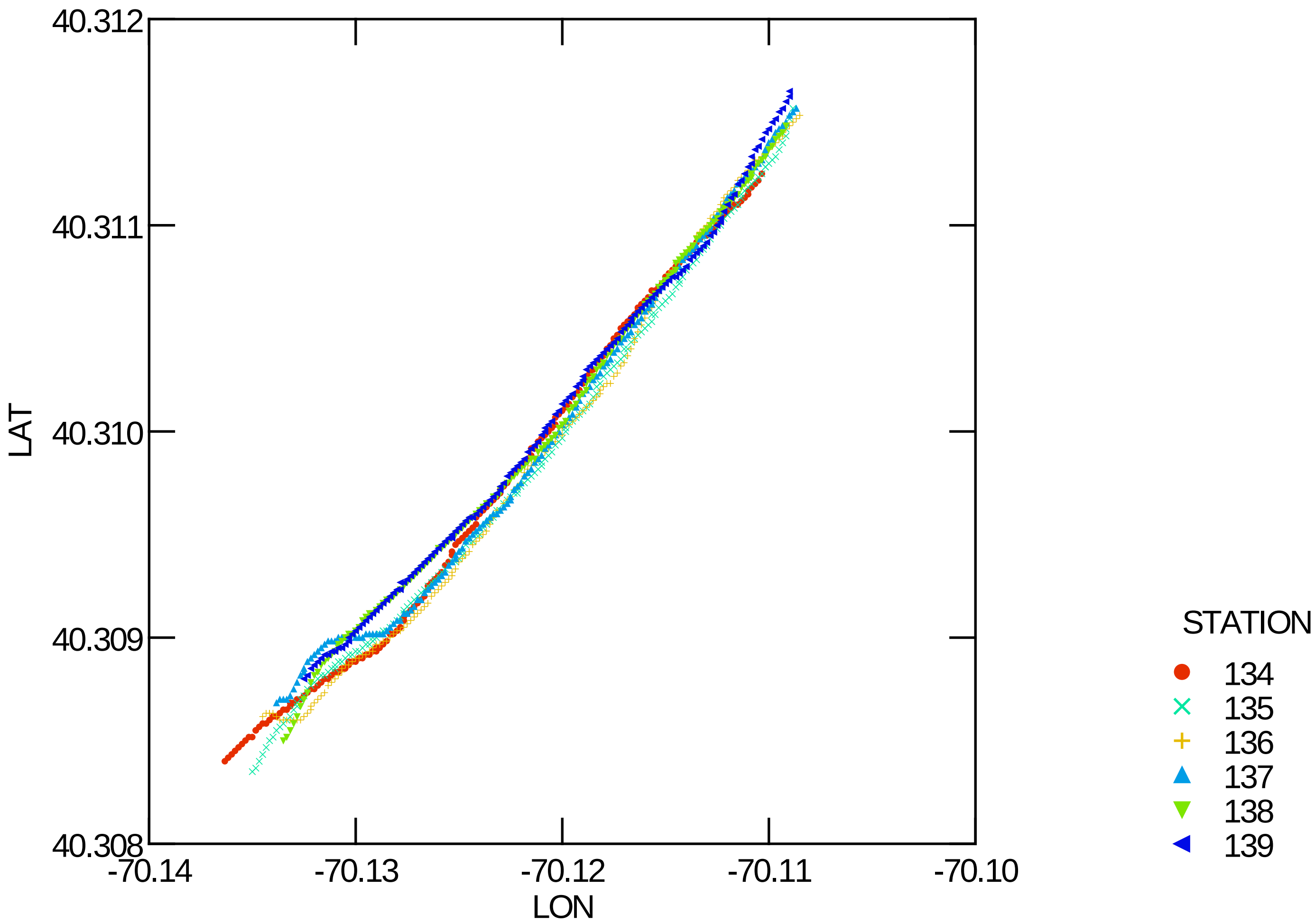

experiment #5
40.317
40.316
40.315
40.314
40.313
40.312
LAT
-70.10
-70.09
-70.08
-70.07
-70.06
LON
STATION
415
416
417
418
419
420

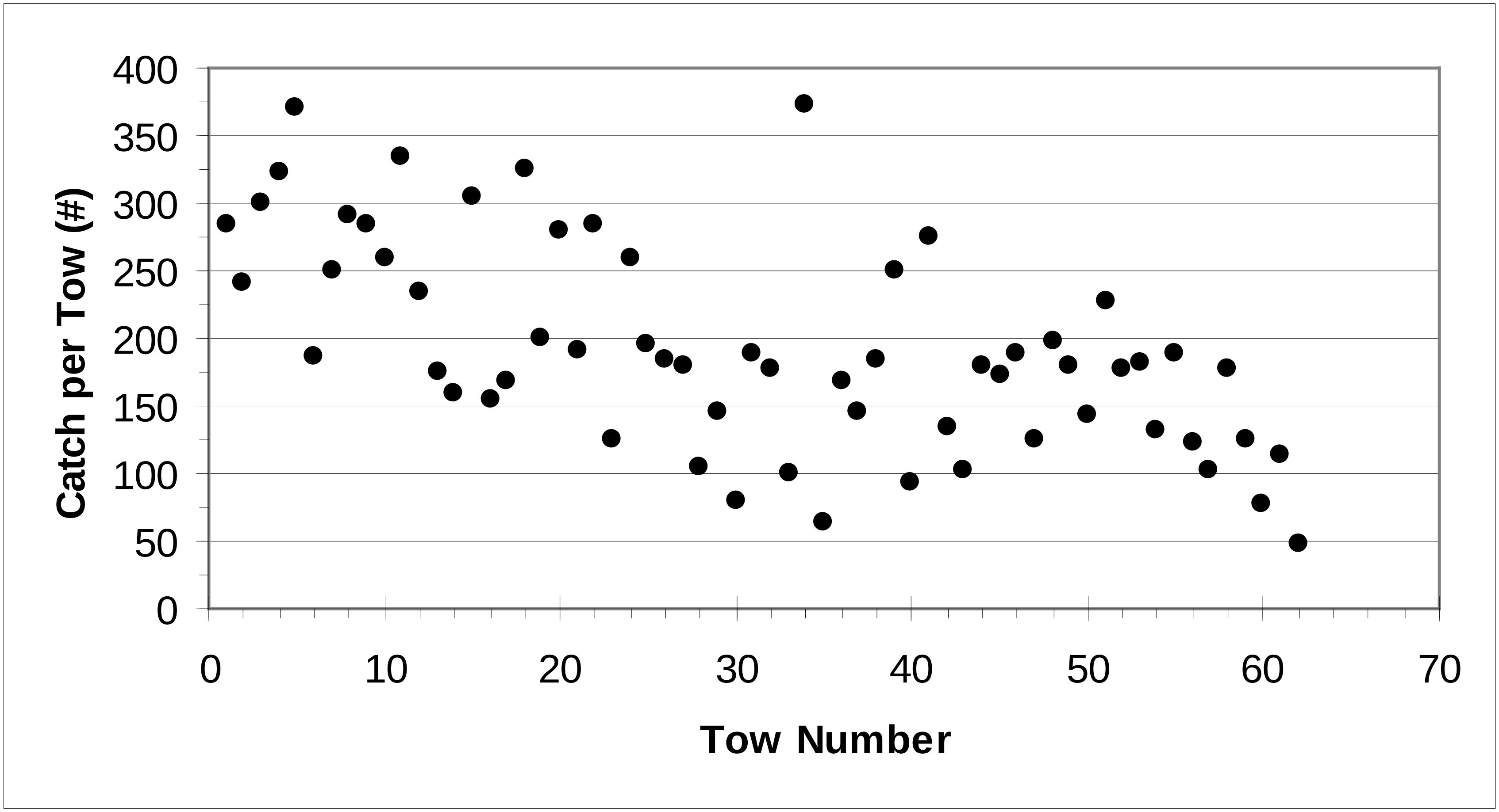

ROW
COL
Count
RCW
CCL
TOW
7
6
5
4
3
2
1
0

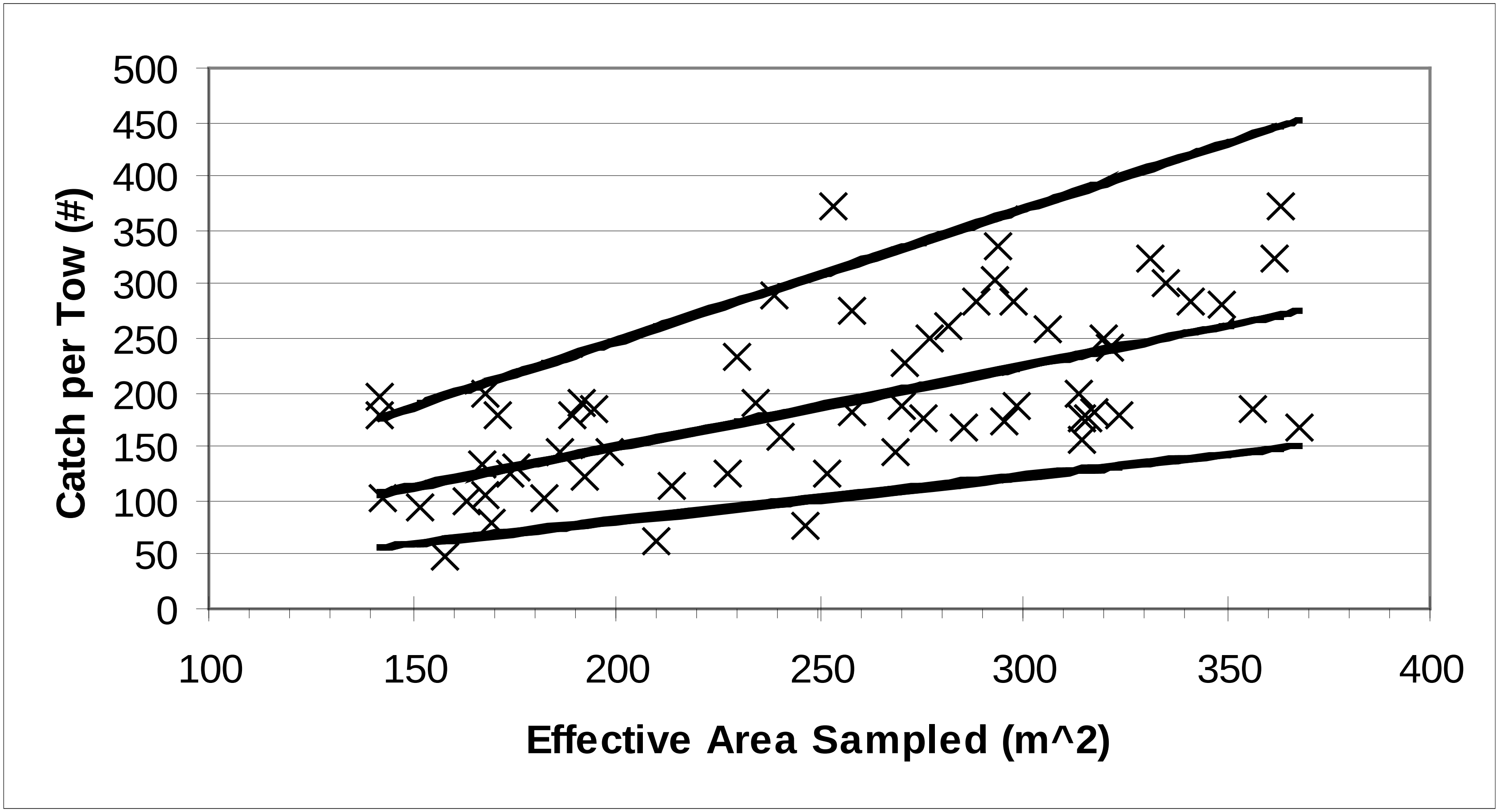
The hit frequency matrix for the Delaware II depletion experiment

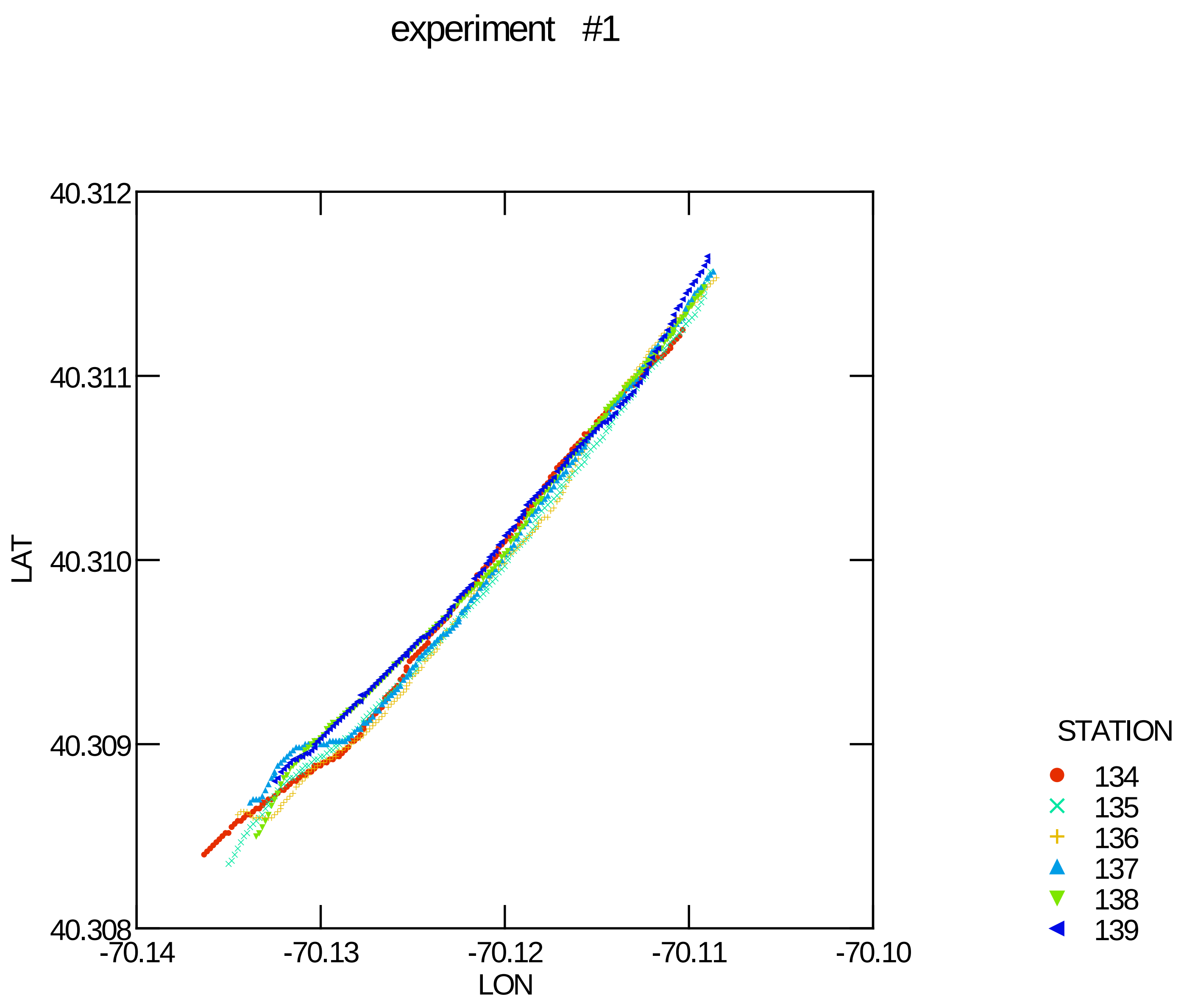


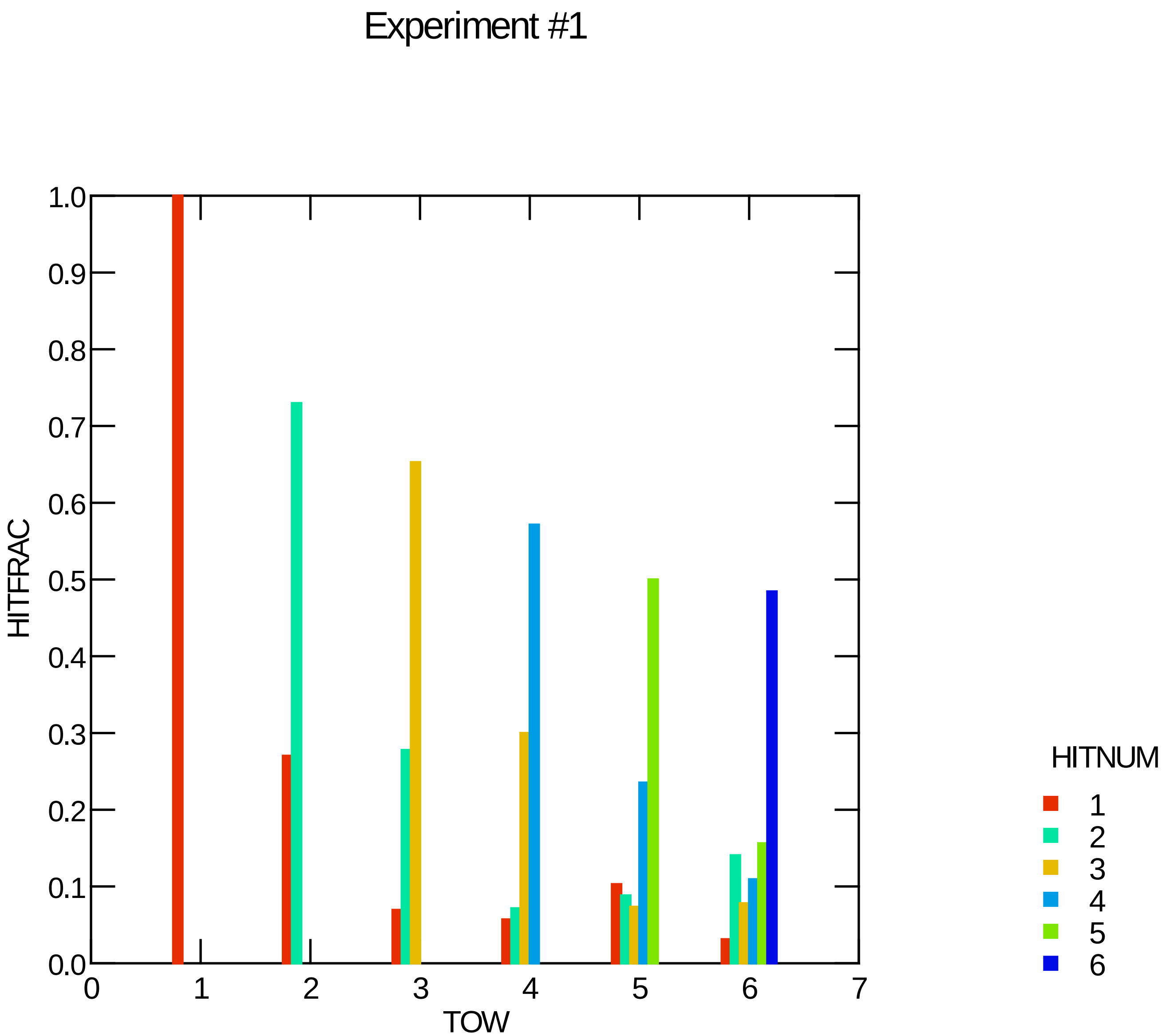
experiment #1

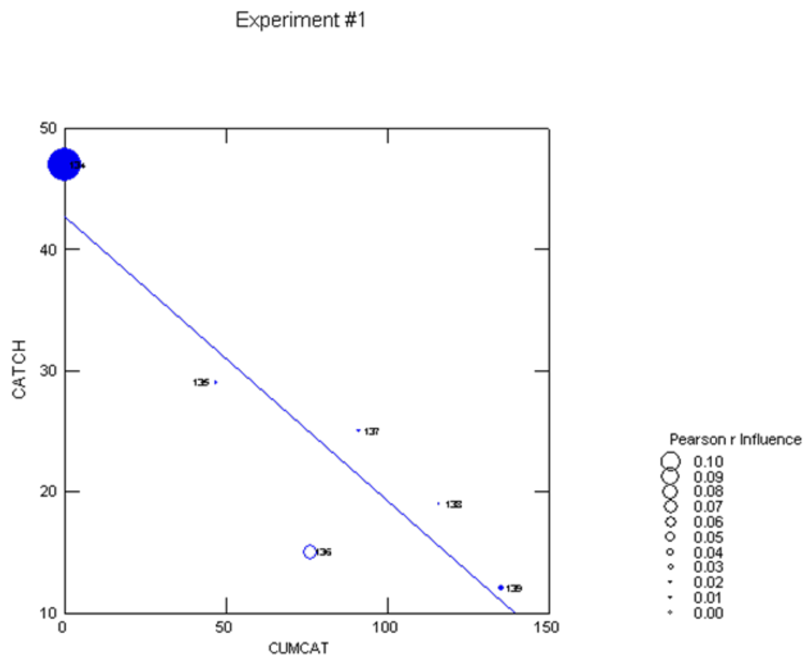



400
350
300
250
200
150
100
50
0
Catch per Tow (#)
0
10
20
30
40
50
60
70
Tow Number


Catch per Tow (#)
500
450
400
350
300
250
200
150
100
50
0
100
150
200
250
300
350
400
Effective Area Sampled (m^2)

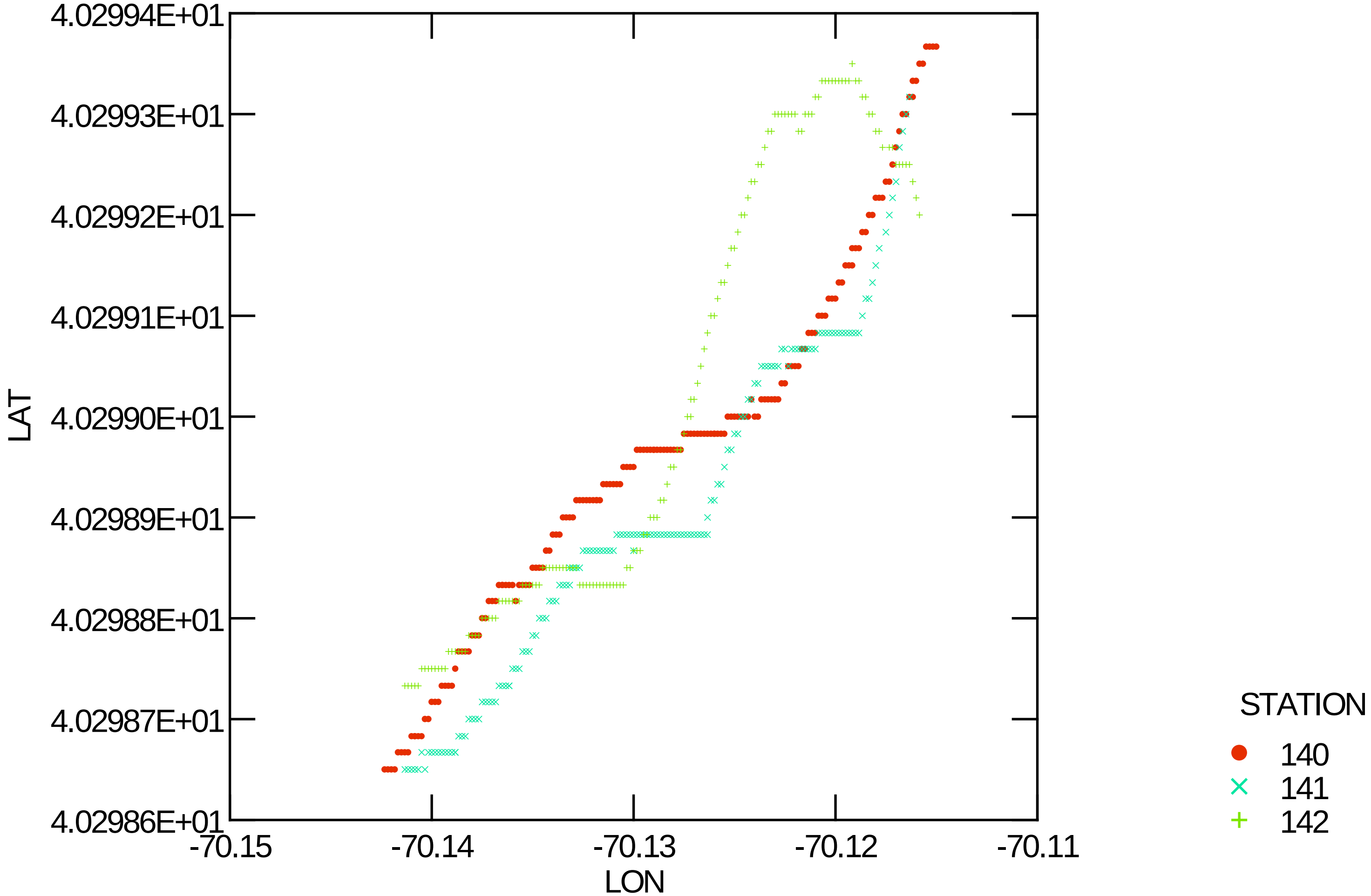

experiment #1
40.312
40.311
40.310
40.309
40.308
LAT
-70.14
-70.13
-70.12
-70.11
-70.10
LON
STATION
134
135
136
137
138
139


Experiment #1
HITFRAC
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
0
1
2
3
4
5
6
7
TOW
HITNUM
1
2
3
4
5
6

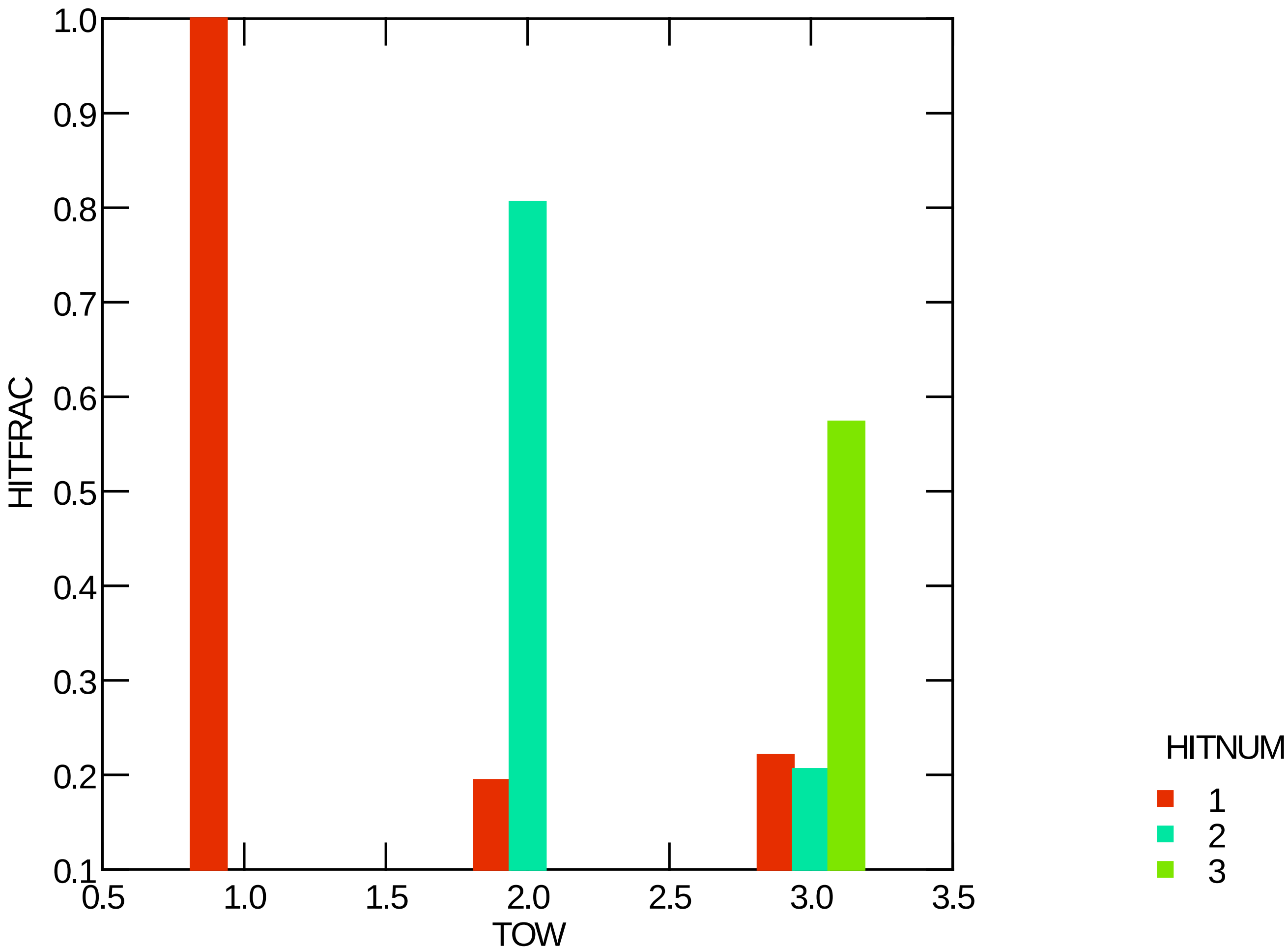


Tow	Area Swept	Efftv Area	C_5%	C_50%	C_pred	OBS	C_95%
CI 1	566598.0	194434.7	29.	41.	41.	47.	56.
CI 2	572557.0	157118.4	23.	34.	33.	29.	46.
CI 3	569490.0	120094.9	17.	26.	25.	15.	36.
CI 4	545629.0	90817.8	12.	20.	19.	25.	29.
CI 5	525474.0	78258.5	10.	17.	16.	19.	25.
CI 6	509265.0	62348.7	8.	14.	13.	12.	21.

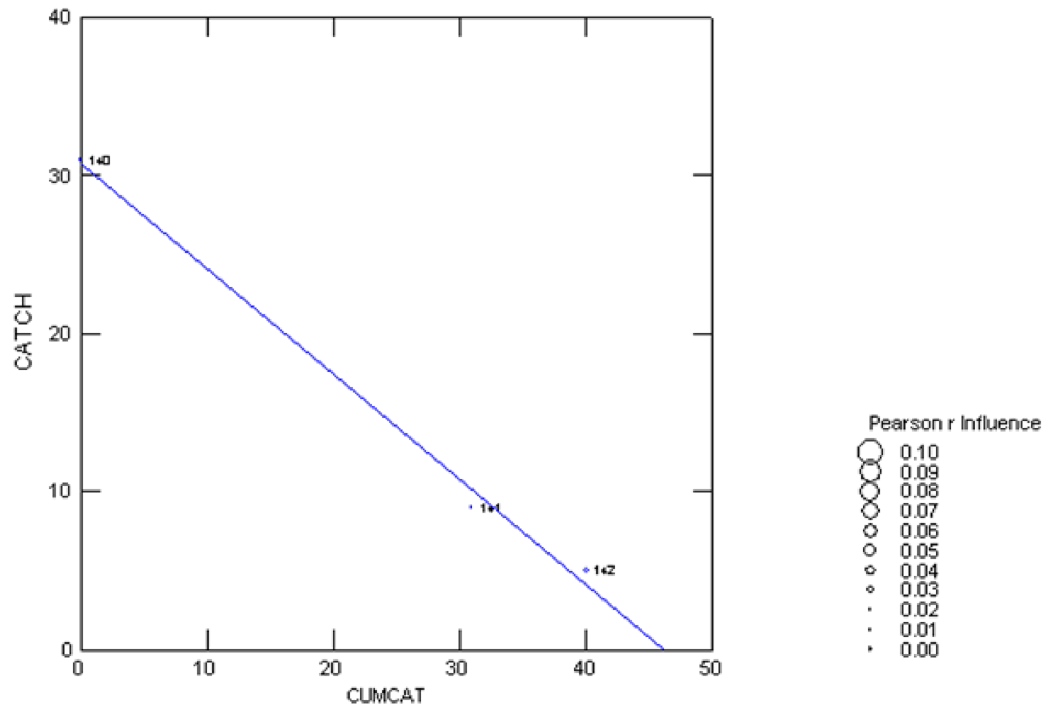
experiment #2



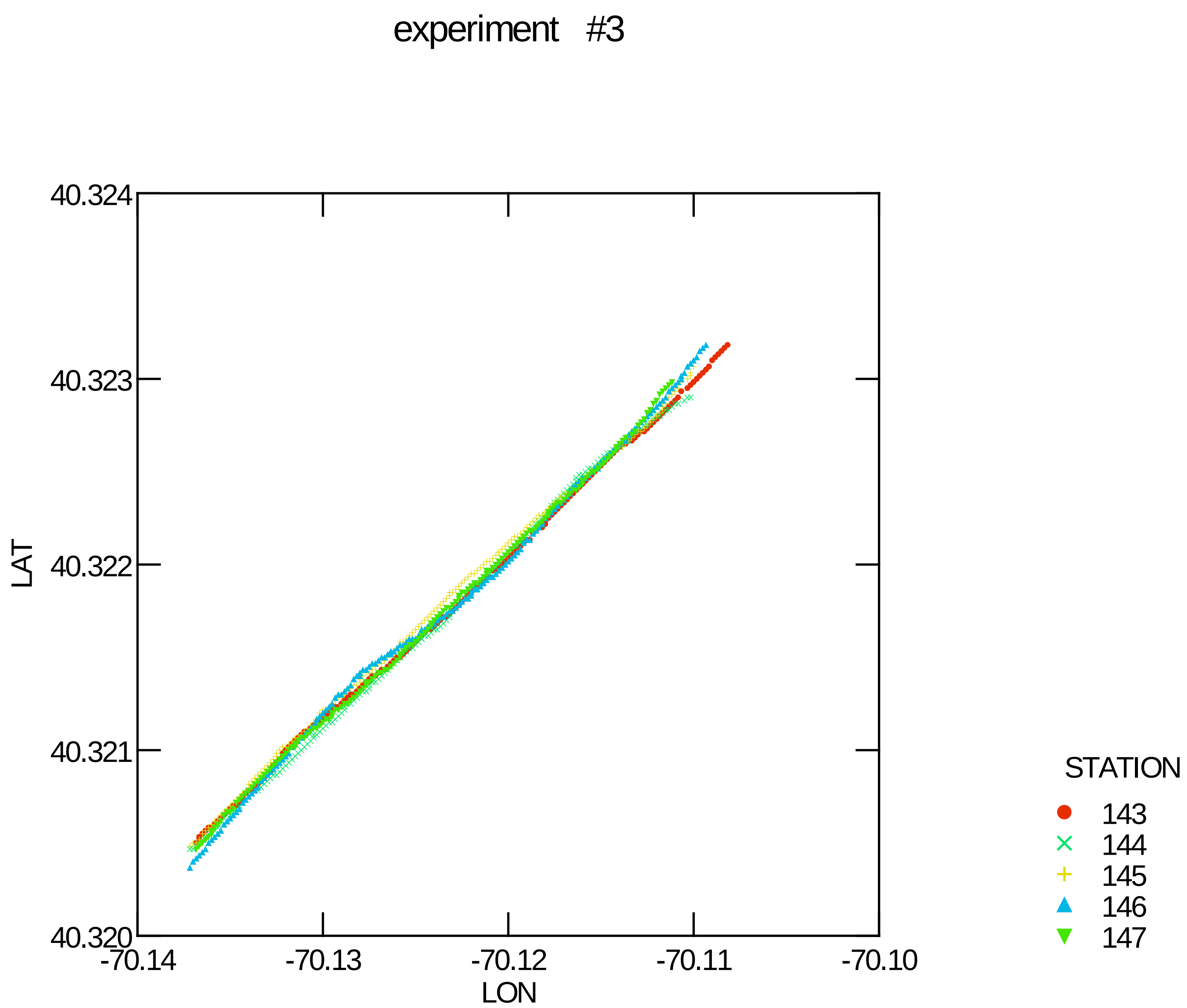
Experiment #2



Experiment #2

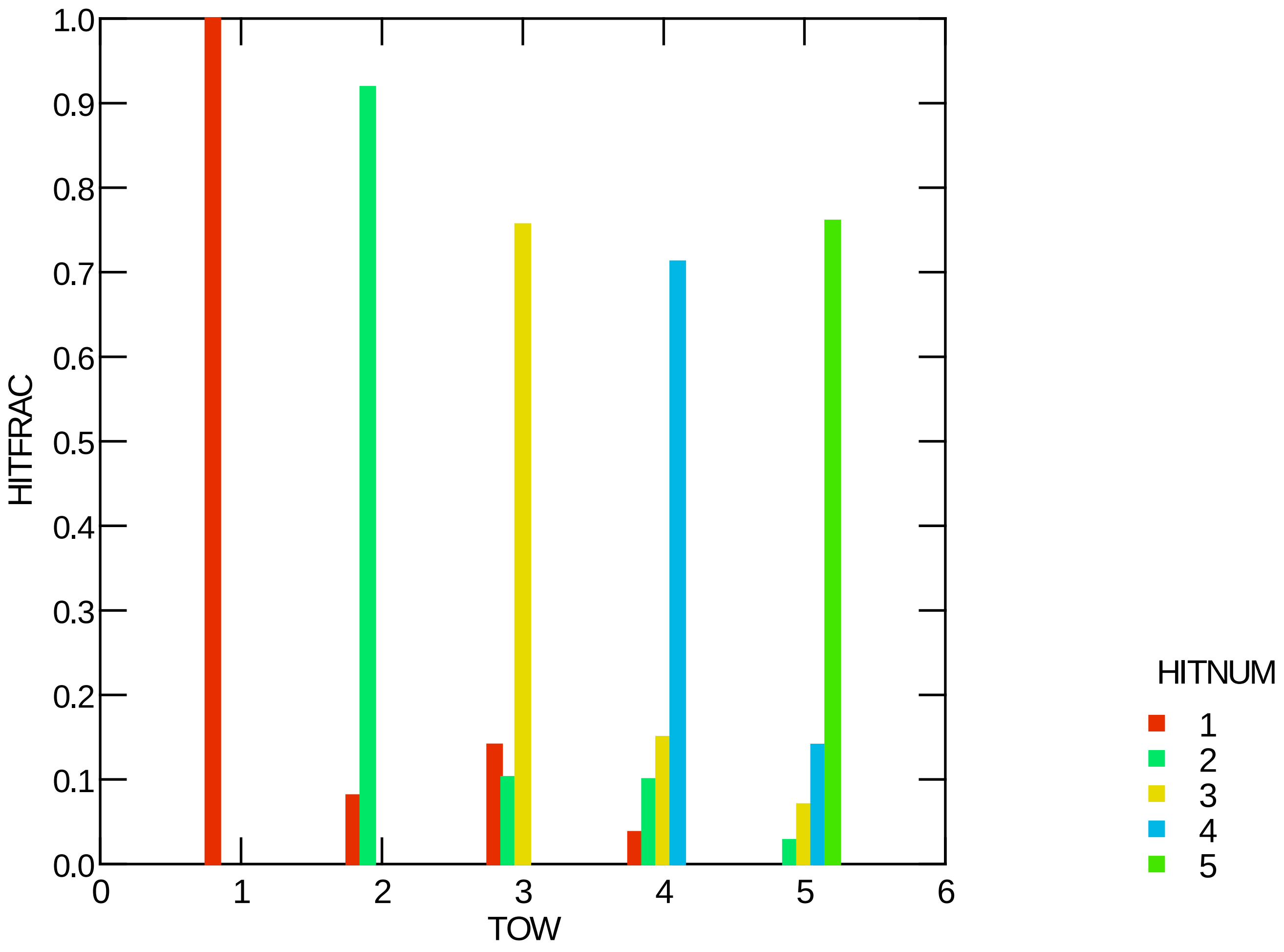


Tow	Area Swept	Efftv Area	C_5%	C_50%	C_pred	OBS	C_95%
CI 1	590158.0	560650.0	20.	28.	28.	31.	38.
CI 2	533897.0	196483.1	6.	10.	10.	9.	16.
CI 3	545551.0	157031.4	4.	9.	8.	5.	14.

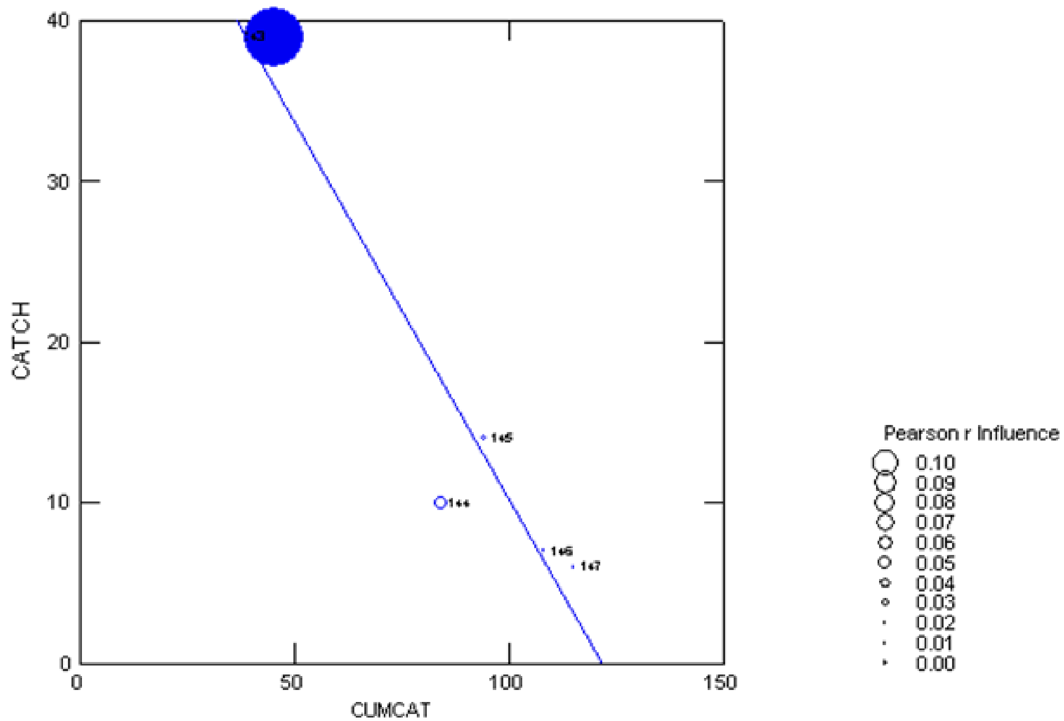


experiment #3
40.324
40.323
40.322
40.321
40.320
LAT
-70.14
-70.13
-70.12
-70.11
-70.10
LON
STATION
143
144
145
146
147

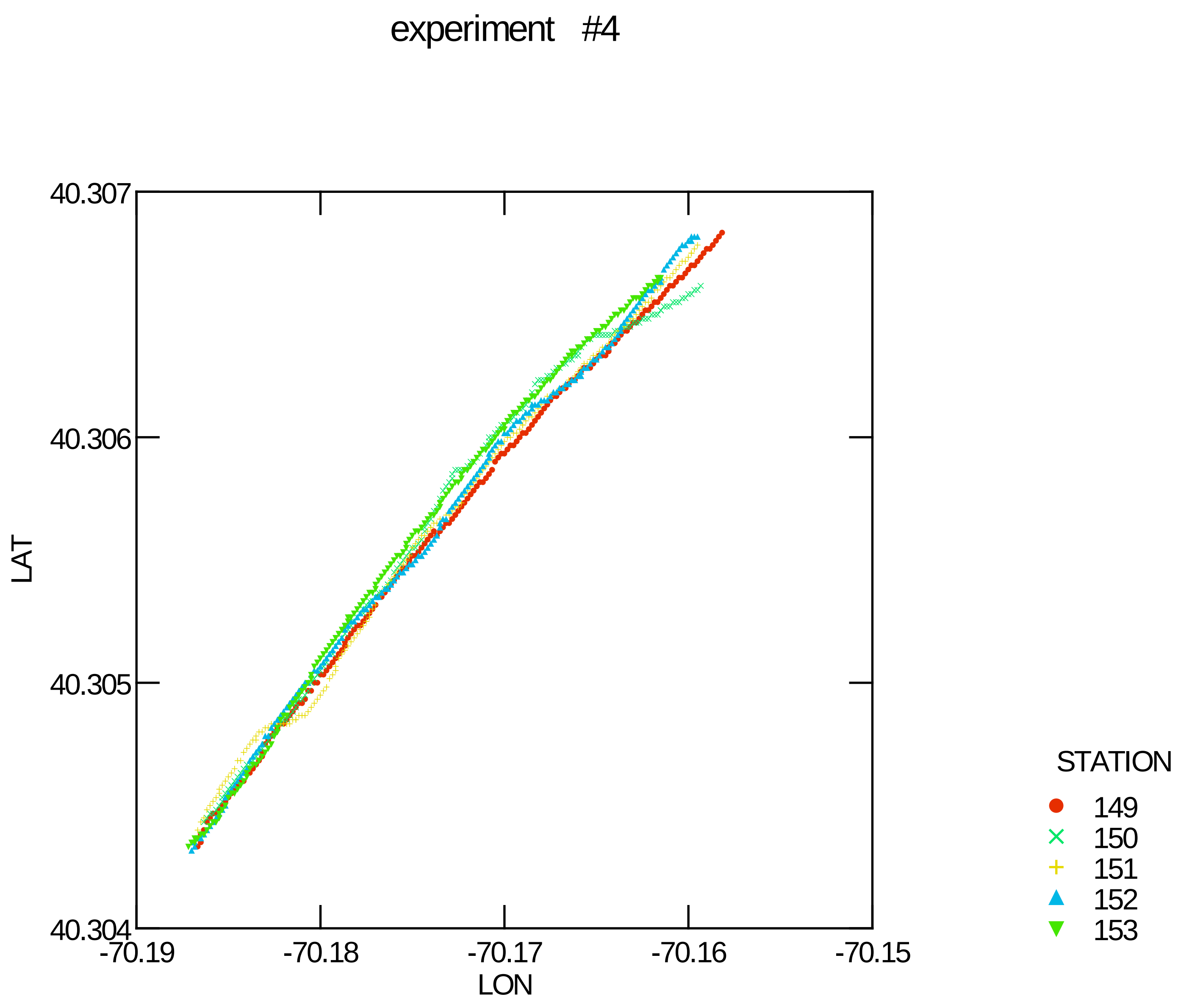
Experiment #3

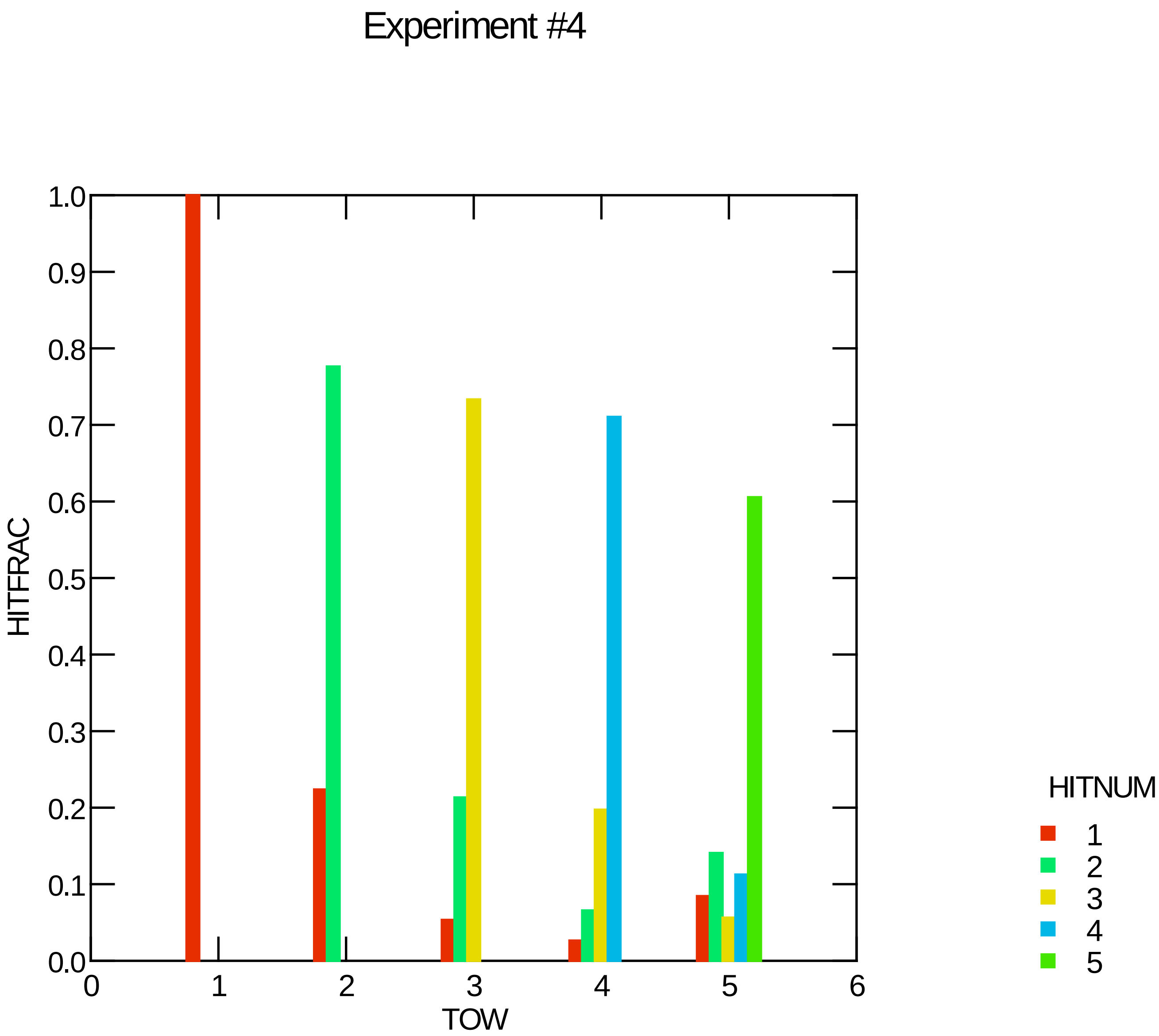


Experiment #3

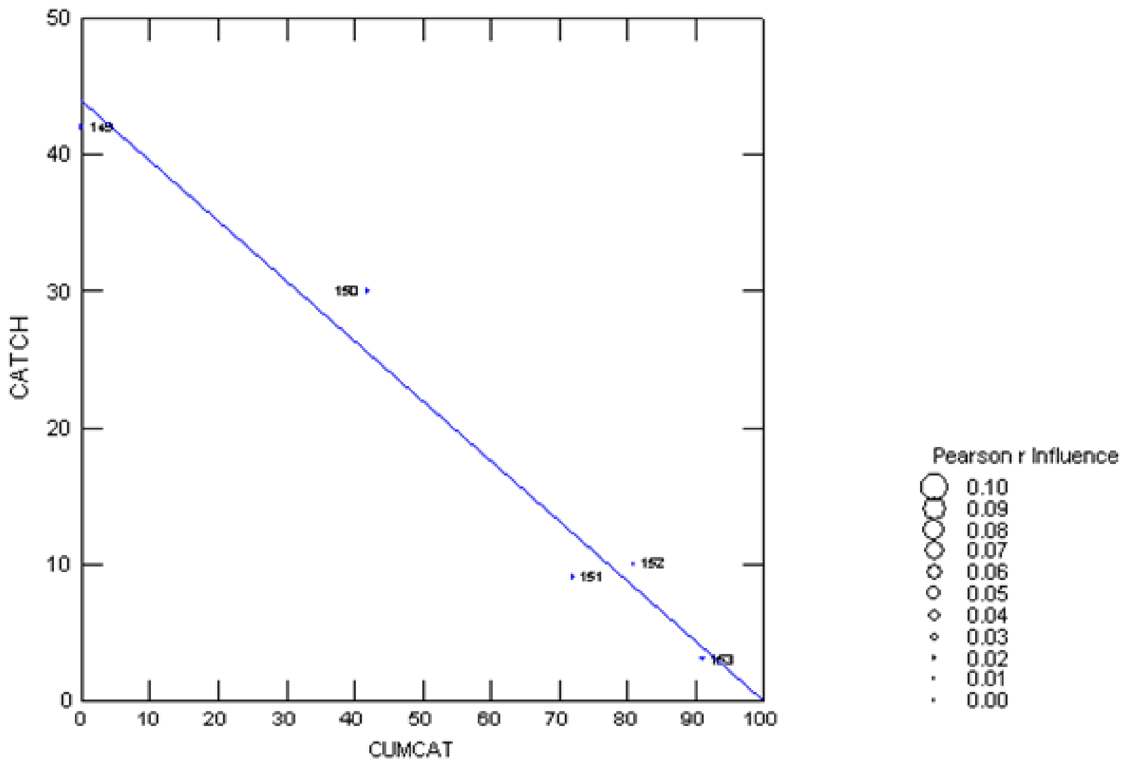


	Tow	Area Swept	Efftv Area	C_5%	C_50%	C_pred	OBS	C_95%
CI	1	634514.0	346094.2	21.	32.	32.	39.	46.
CI	2	590274.0	192848.7	11.	18.	18.	10.	28.
CI	3	597123.0	143012.4	8.	14.	13.	14.	22.
CI	4	612709.0	89918.2	5.	9.	8.	7.	15.
CI	5	562786.0	43035.3	2.	5.	4.	6.	9.

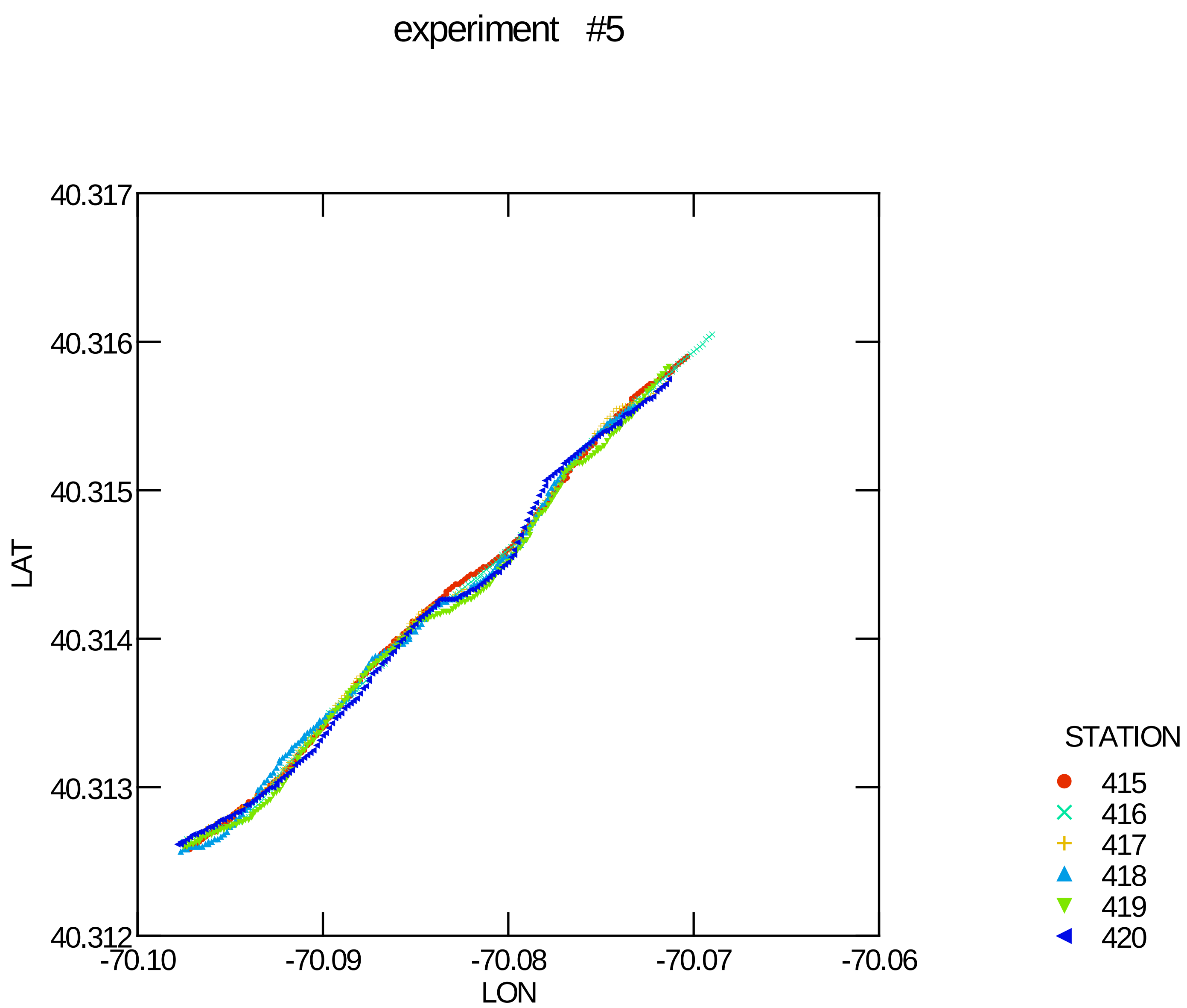

experiment #4
40.307
40.306
40.305
40.304
LAT
-70.19
-70.18
-70.17
-70.16
-70.15
LON
STATION
149
150
151
152
153


Experiment #4
HITFRAC
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
0
1
2
3
4
5
6
TOW
HITNUM
1
2
3
4
5

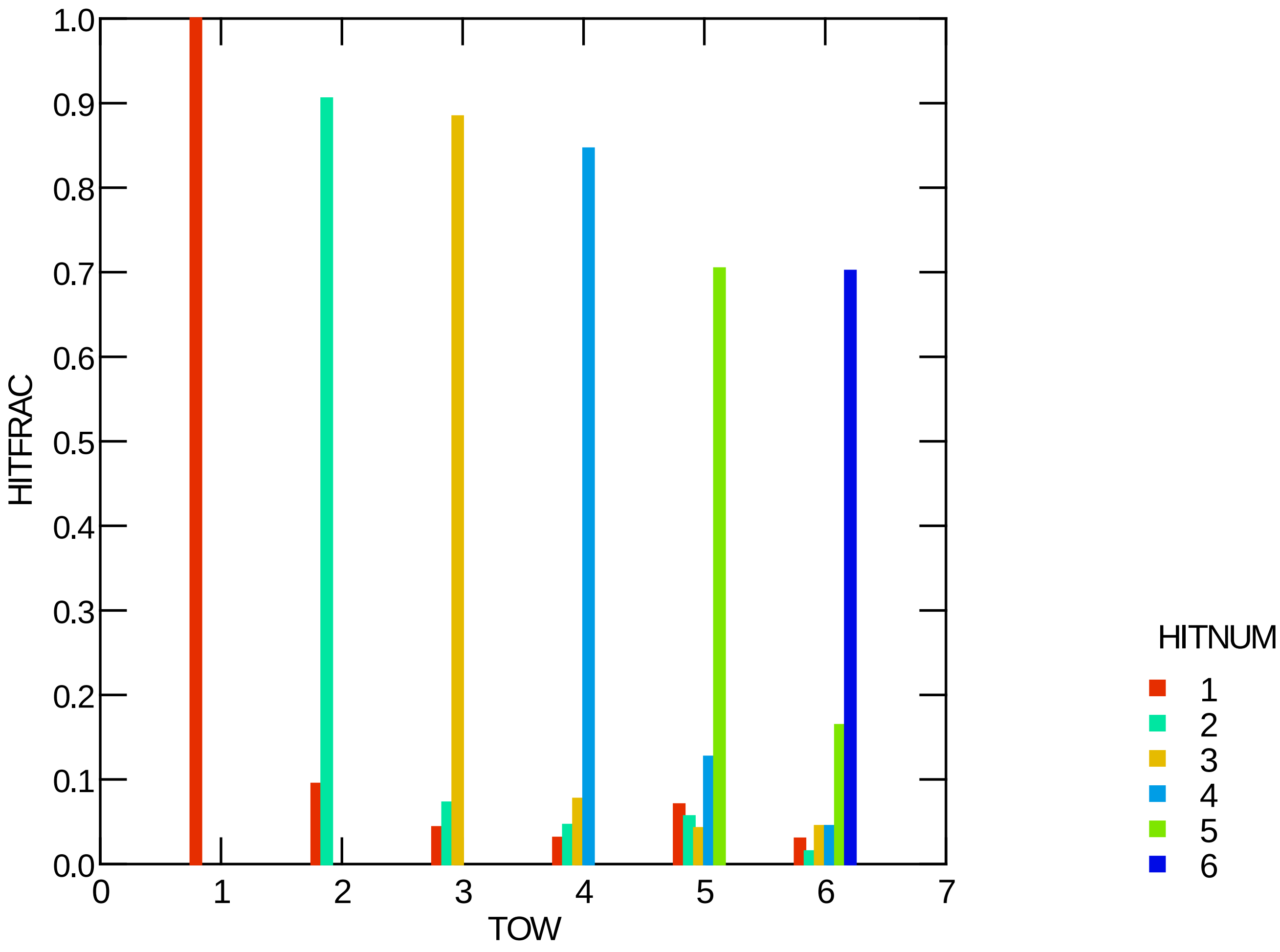
Experiment #4



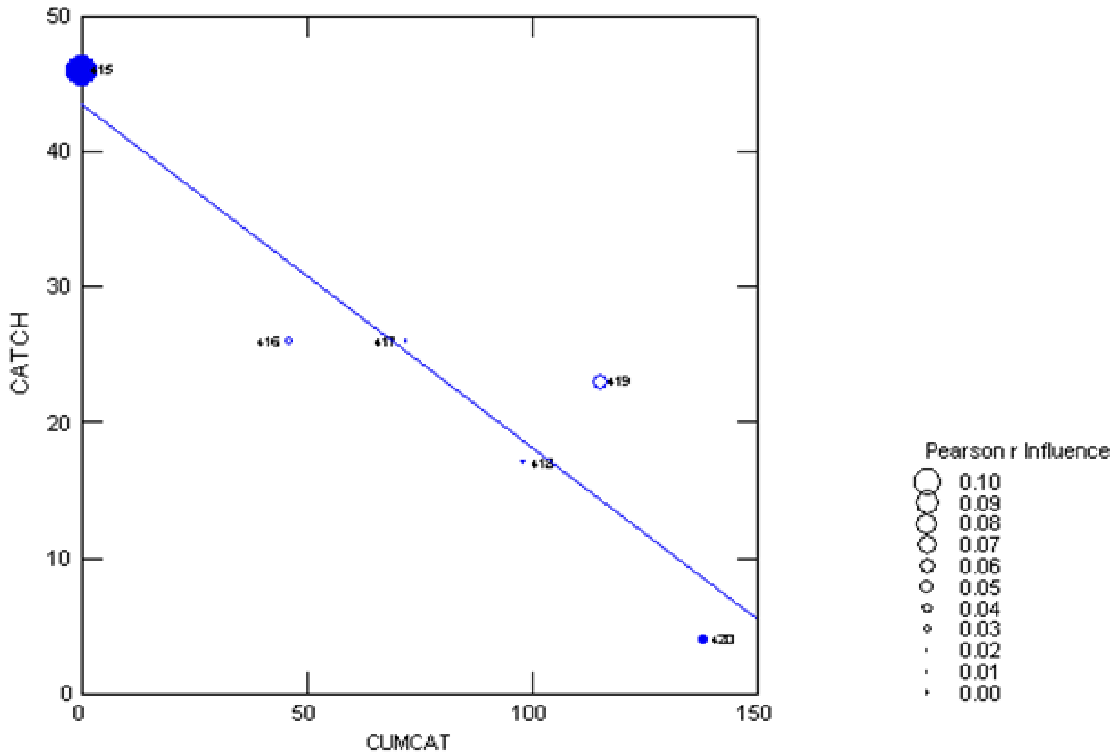
Tow	Area Swept	Efftv Area	C_5%	C_50%	C_pred	OBS	C_95%
CI 1	623017.0	425046.3	34.	44.	43.	42.	56.
CI 2	586683.0	230645.1	17.	24.	24.	30.	33.
CI 3	592784.0	121917.5	8.	13.	12.	9.	20.
CI 4	603661.0	67315.5	4.	8.	7.	10.	12.
CI 5	561273.0	75208.0	4.	9.	8.	3.	14.


experiment #5
40.317
40.316
40.315
40.314
40.313
40.312
LAT
-70.10
-70.09
-70.08
-70.07
-70.06
LON
STATION
415
416
417
418
419
420

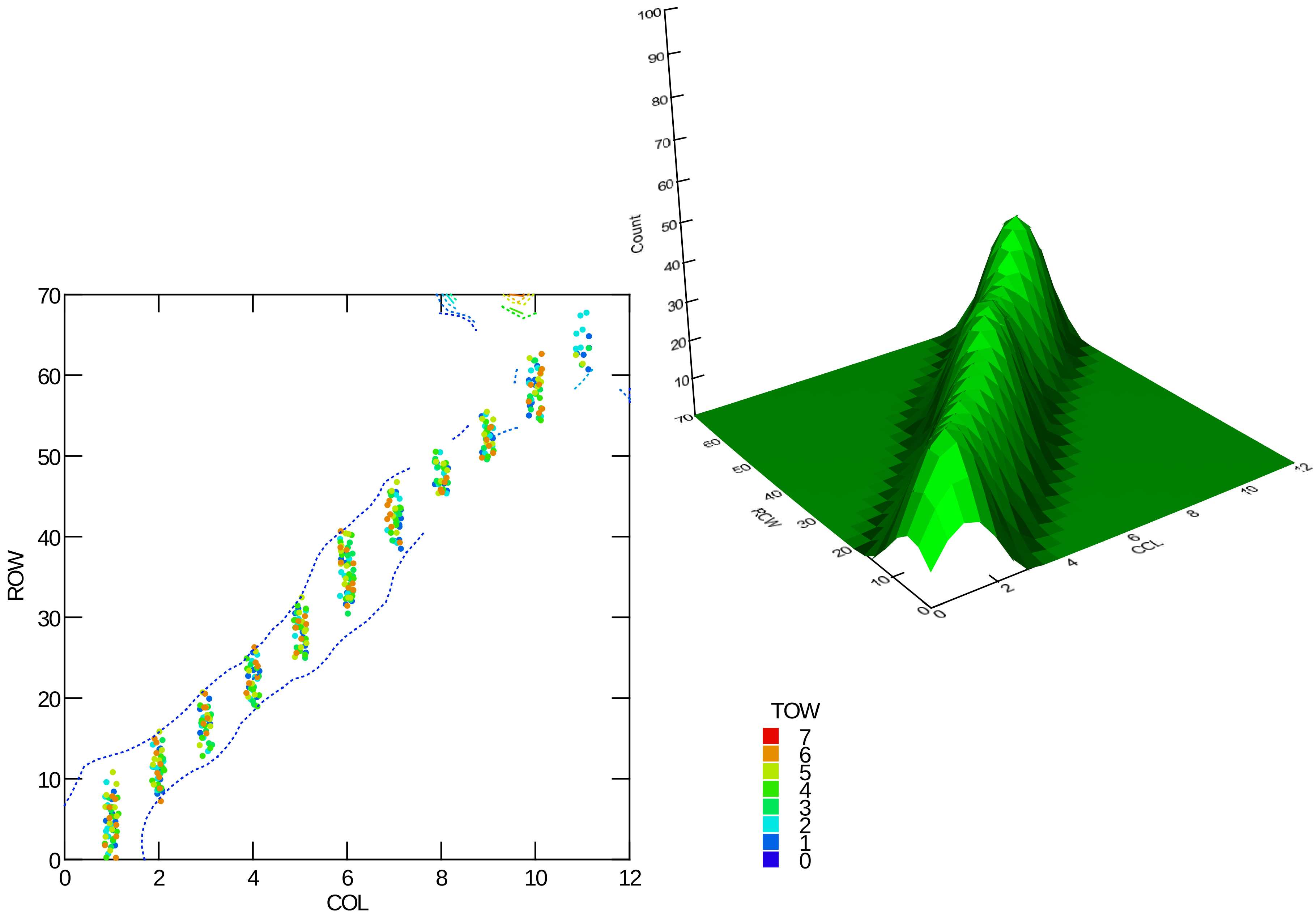
Experiment #5



Experiment #5



Tow	Area Swept	Efftv Area	C_5%	C_50%	C_pred	OBS	C_95%
CI 1	682690.0	261011.9	30.	46.	45.	46.	65.
CI 2	724422.0	200259.1	23.	35.	35.	26.	51.
CI 3	655756.0	130296.5	14.	23.	23.	26.	35.
CI 4	615450.0	90080.8	9.	16.	16.	17.	25.
CI 5	658380.0	84510.7	9.	15.	15.	23.	24.
CI 6	667457.0	58153.5	6.	11.	10.	4.	18.


70
60
50
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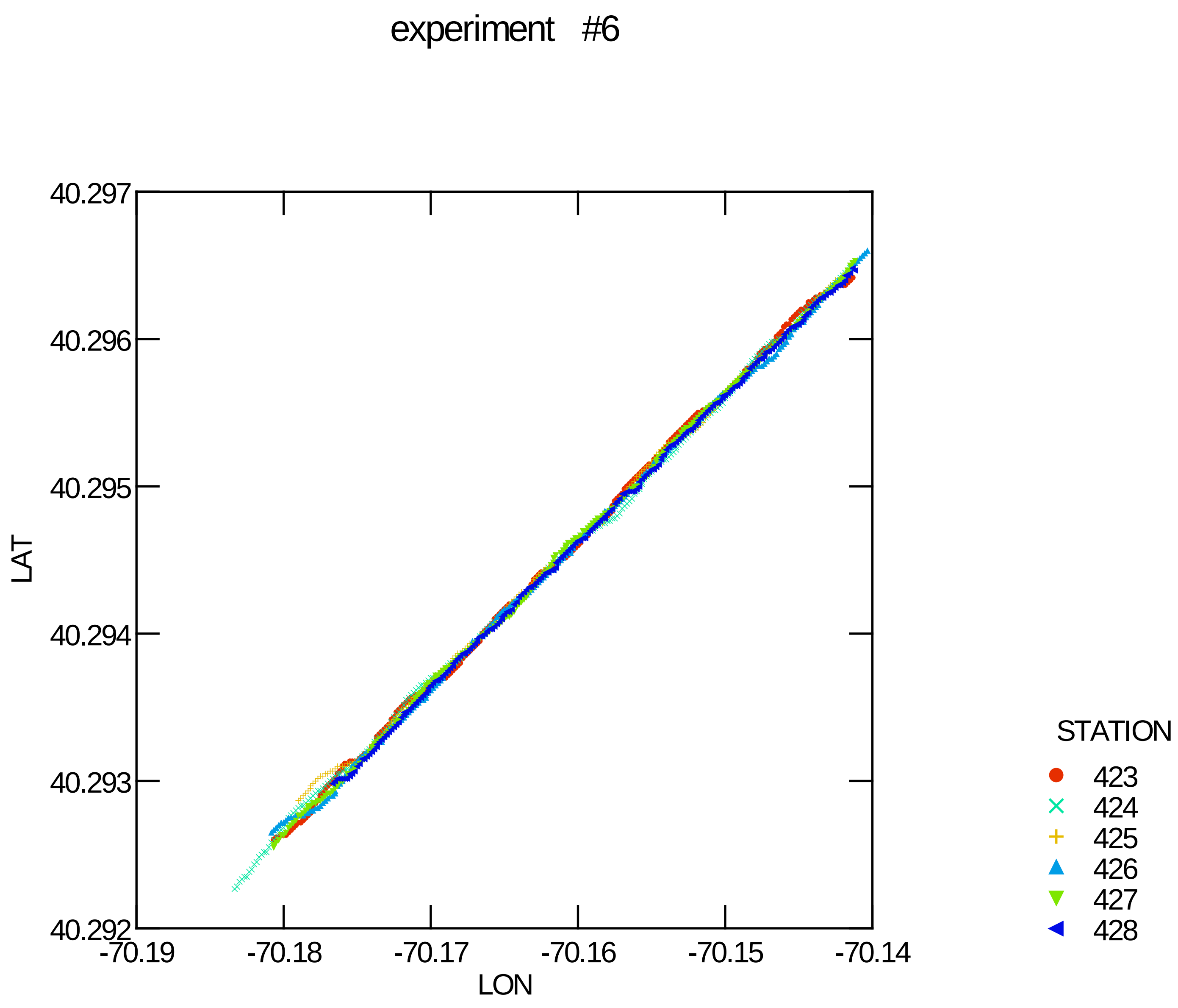
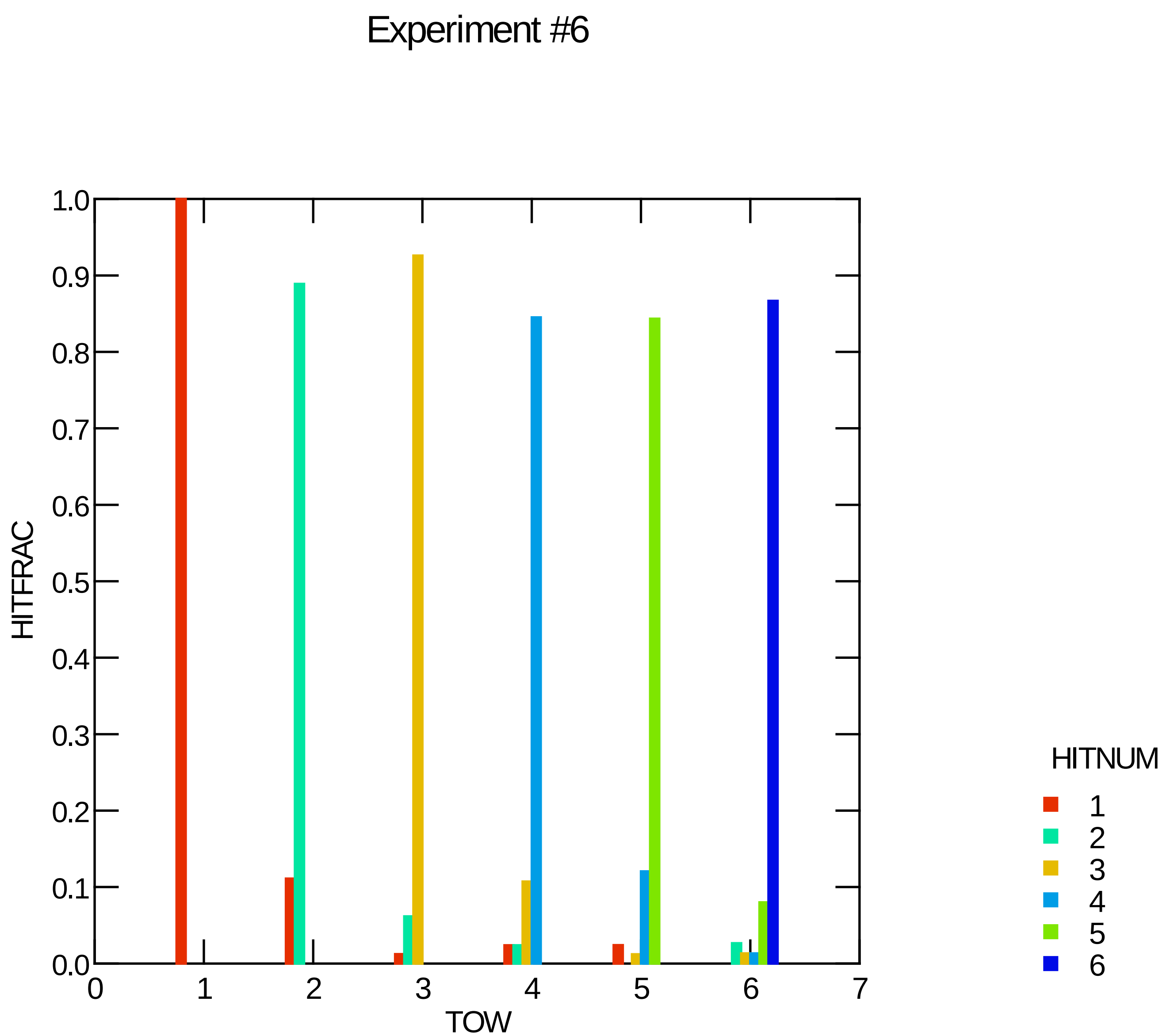
30
20
10
0
0
2
4
6
8
10
12
ROW
COL
Count
100
90
80
70
60
50
40
30
20
10
TOW
7
6
5
4
3
2
1
0

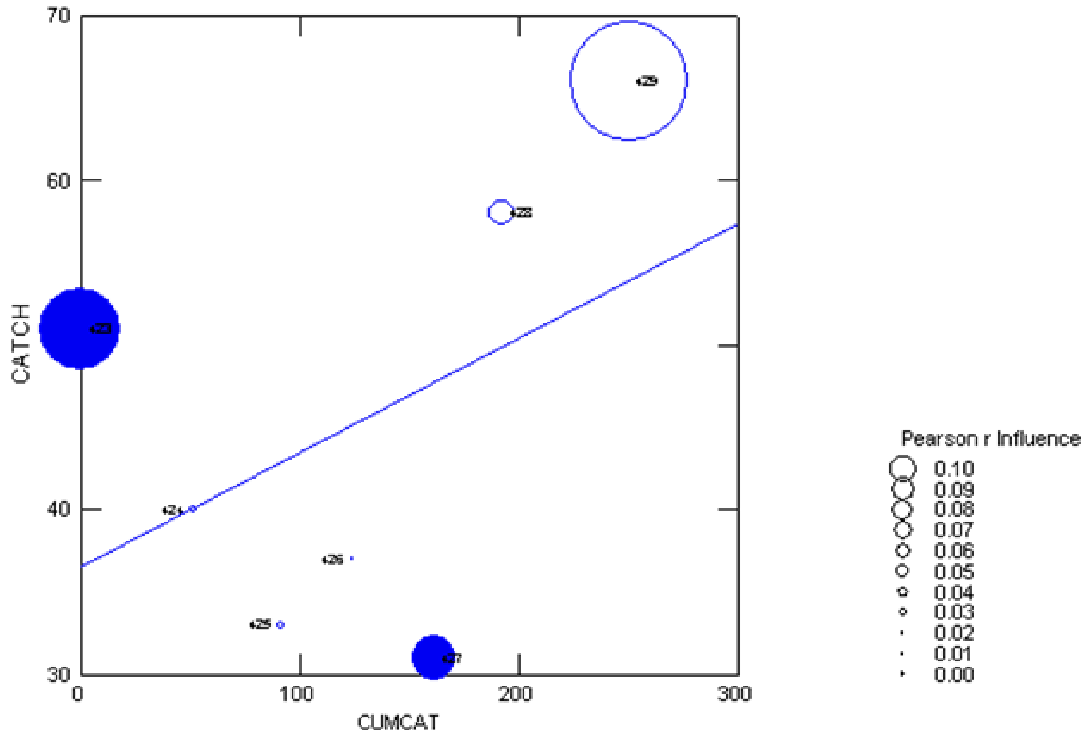


experiment #6
40.297
40.296
40.295
40.294
40.293
40.292
LAT
-70.19
-70.18
-70.17
-70.16
-70.15
-70.14
LON
STATION
423
424
425
426
427
428



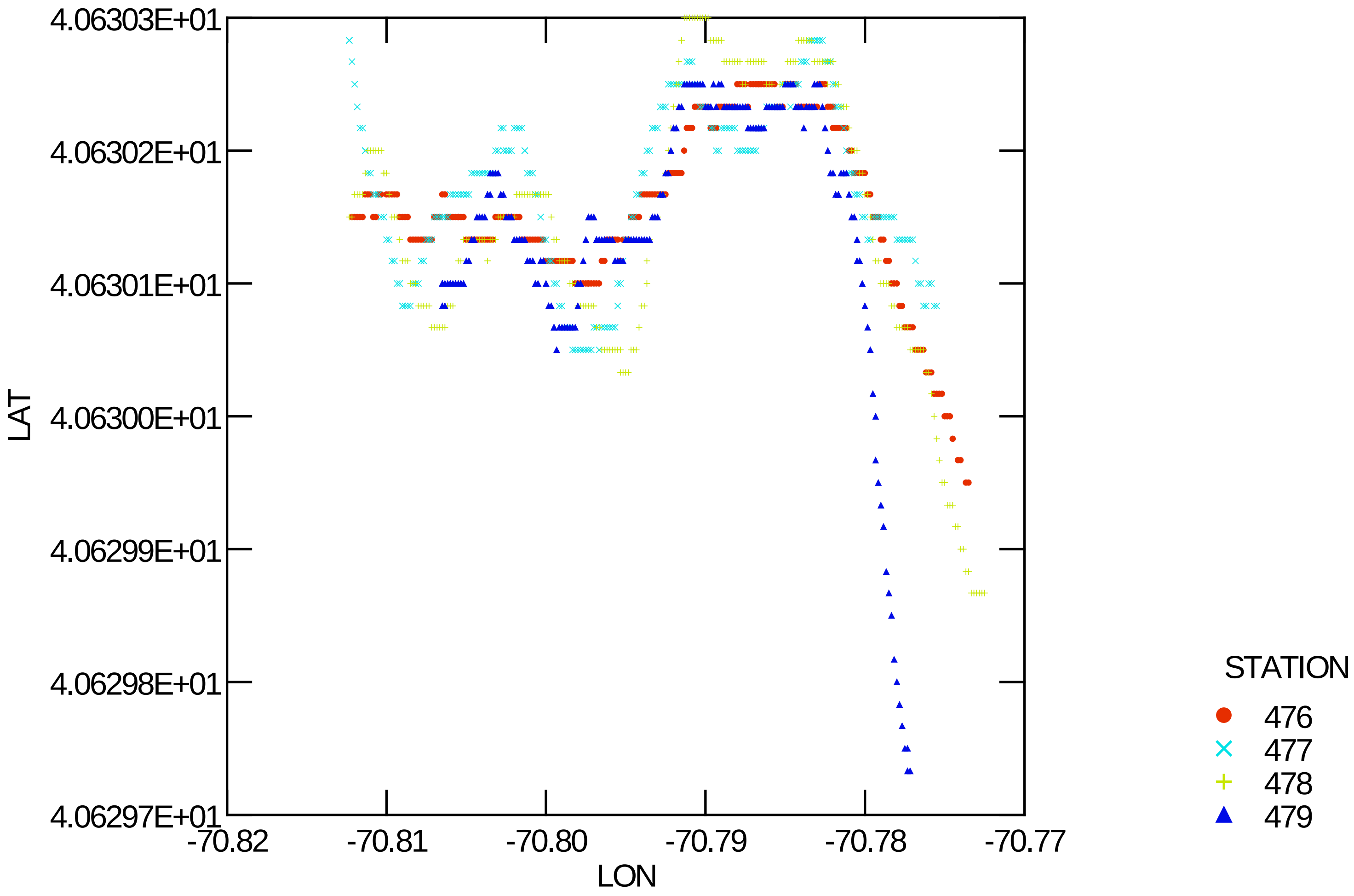
Experiment #6
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
HITFRAC
0
1
2
3
4
5
6
7
TOW
HITNUM
1
2
3
4
5
6

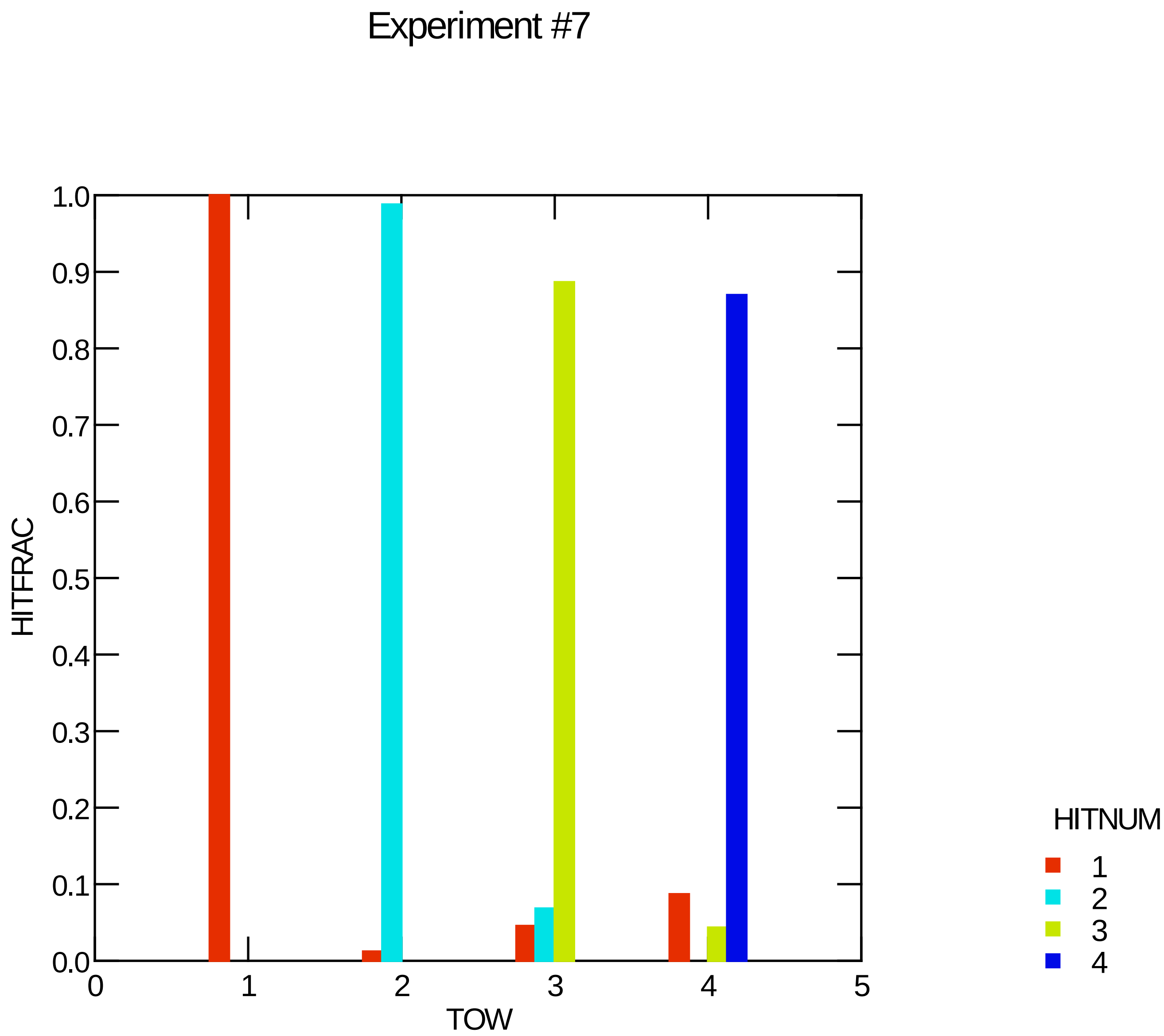
Experiment #6

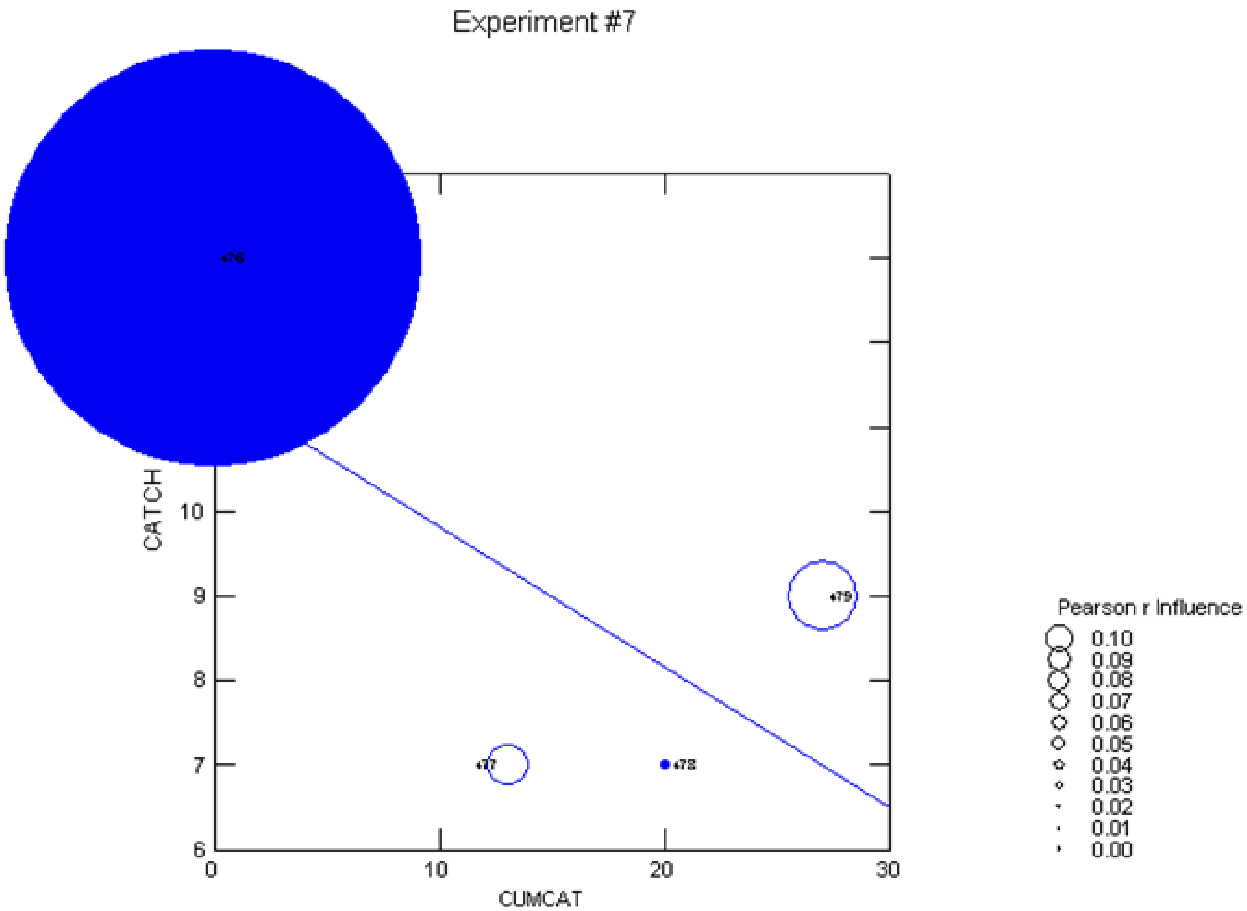


	Tow	Area Swept	Efftv Area	C_5%	C_50%	C_pred	OBS	C_95%
CI	1	1149328.0	57466.9	28.	47.	47.	51.	71.
CI	2	1228838.0	59257.8	29.	48.	48.	40.	73.
CI	3	1101188.0	50926.9	25.	41.	41.	33.	63.
CI	4	1180802.0	52736.3	26.	43.	43.	37.	65.
CI	5	1158107.0	49689.5	24.	40.	40.	31.	62.
CI	6	1031628.0	42518.2	20.	35.	34.	58.	53.

experiment #7

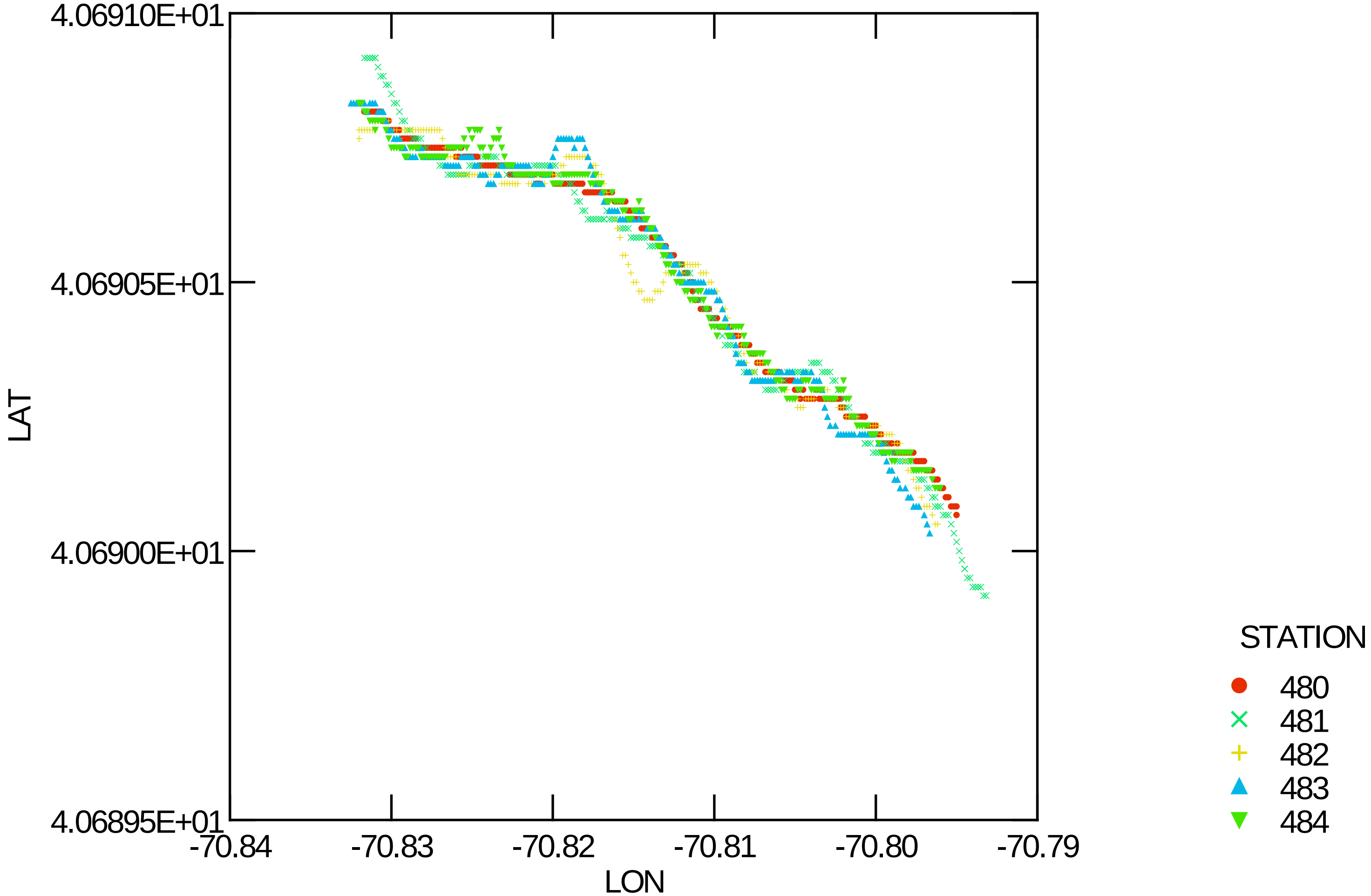



Experiment #7
HITFRAC
1.0
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0.0
0
1
2
3
4
5
TOW
HITNUM
1
2
3
4

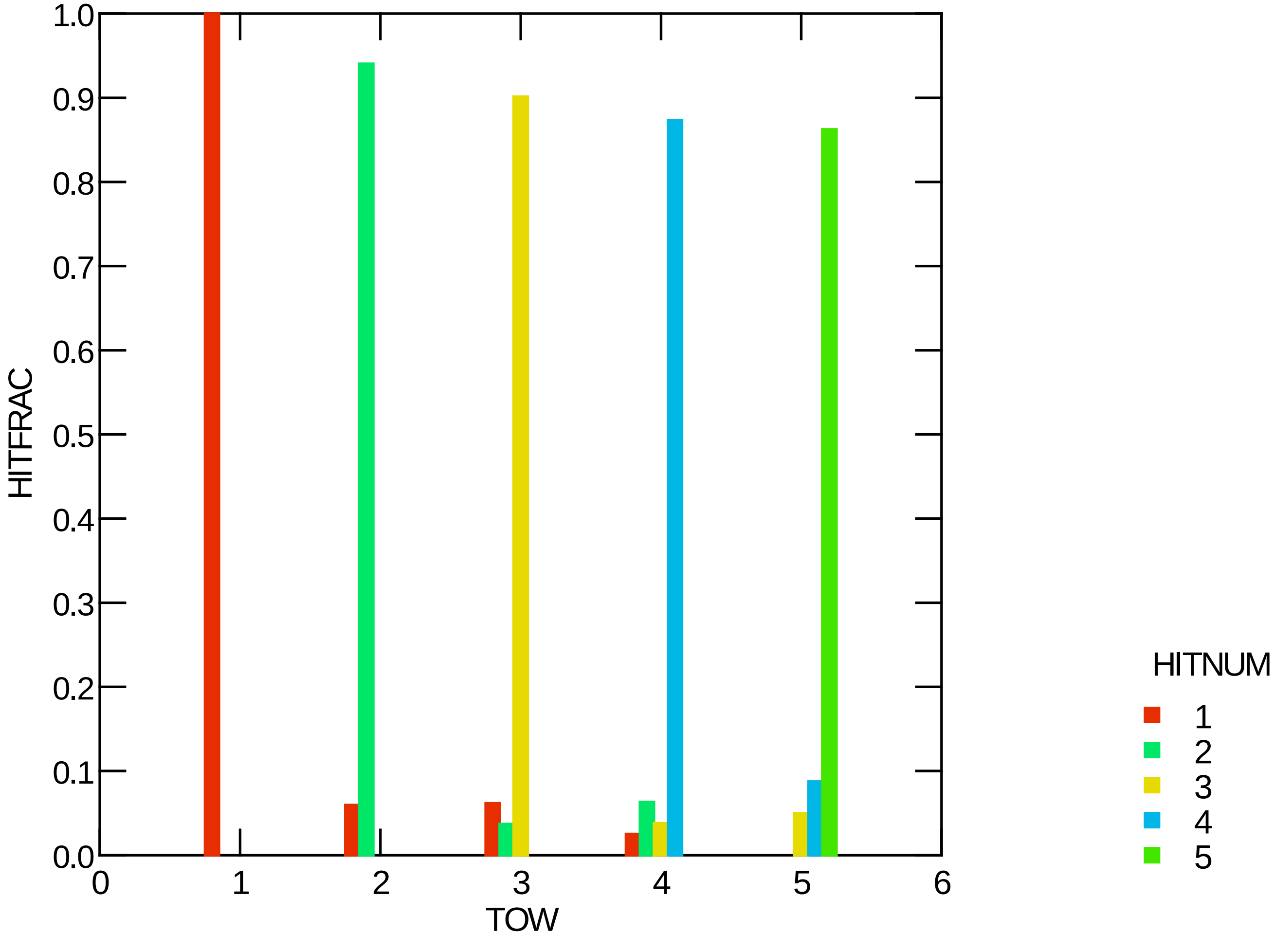


	Tow	Area Swept	Efftv Area	C_5%	C_50%	C_pred	OBS	C_95%
CI	1	928448.0	107975.1	7.	12.	11.	13.	17.
CI	2	894369.0	94451.2	6.	10.	9.	7.	16.
CI	3	966692.0	94029.9	6.	10.	9.	7.	16.
CI	4	719680.0	64567.9	4.	7.	6.	9.	12.

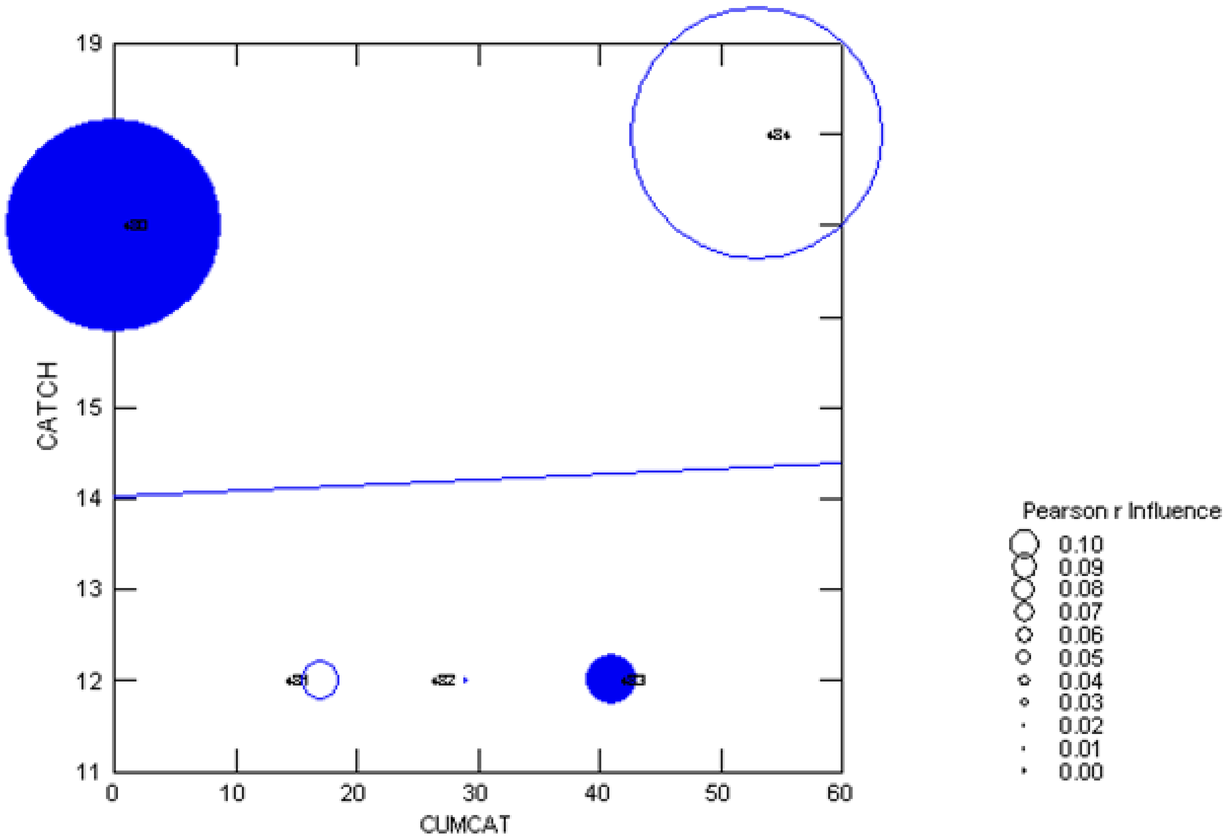
experiment #8



Experiment #8



Experiment #8



Tow		Area Swept	Efftv Area	C_5%	C_50%	C_pred	OBS	C_95%
CI	1	926410.0	46320.6	10.	16.	15.	17.	23.
CI	2	969823.0	46667.1	10.	16.	16.	12.	23.
CI	3	899735.0	41741.6	9.	15.	14.	12.	21.
CI	4	891727.0	39852.1	9.	14.	13.	12.	21.
CI	5	904538.0	38716.8	8.	14.	13.	18.	20.

Appendix A1b: Analysis of Depletion Experiments

DISCUSSION DOCUMENT FOR SARC
APRIL 12, 2010

This information is distributed solely for the purpose of pre-dissemination peer review. It has not been formally disseminated by NOAA. It does not represent any final agency determination or policy.

Prepared by Paul Rago

F/V Mary K Depletion Exp#1- Monkfish Cooperative Survey 2009 Cookie sweep

```
 Initial Values of parameters
    .00009               =Mean  density of monkfish per sq ft
    .60000               =Efficiency of trawl
 800.00000               =K parameter for negative binomial dist
    .79900              =Gamma parameter;  initial guess=trawl width/cell width
 Bounds on parameters
  Param #            Lower Bound                Upper Bound
          1   1.000000000000000E-007   2.000000000000000E-002
          2   5.000000000000000E-002   9.500000000000000E-001
          3   5.000000000000000E-001     2000.100000000000000
          4   8.000000000000000E-001   8.001000000000000E-001

 max # tows=          6
 Max number of hits =           6

 Value of likelihood function at initial guess:
     .000090     .600000  800.000000     .799000       44.0247337183

  Starting Pt    Density  Efficiency      K Parameter   Gamma Par  Likelihood Fc
 n
       Init Cond     .000214     .327869     .500000     .800000       27.252829
 Restart from IC     .000214     .327869     .500000     .800000       27.252829
 At 0.75 Current     .000214     .327611     .500000     .800002       27.252829
 At 1.25 Current     .000209     .342985   61.461972     .800001       18.847179
      At Current     .000209     .343162   60.701874     .800002       18.847125
 At 1.25 IniCond     .000215     .327795     .500000     .800077       27.252829
      At Current     .000215     .327795     .500000     .800077       27.252829
 At 0.75 IniCond     .000214     .327697     .500000     .800100       27.252829
      At Current     .000214     .327697     .500000     .800100       27.252829

BEST soln=      .00021      .34316    60.70187      .80000   18.8471245429
   Ave Density/ft^2        .0002091
   Efficiency              .3431617
   K Parameter            60.70187
   Gamma Parameter           .80000

Profile range for m=         .0001715726         .0002532903
Profile range for e=         .2555851981         .4721314370
Profile range for k=   -10784.9992906401     -164.6256330889
Profile range for g=         .5432191982        1.0672223640

 Profile likelihood for Gamma:
                index   gamma   LogLikelihood
  Gamma Profile:    1   .50000       18.86886
  Gamma Profile:    2   .51250       18.87012
  Gamma Profile:    3   .52500       18.86543
  Gamma Profile:    4   .53750       18.86469
  Gamma Profile:    5   .55000       18.86293
  Gamma Profile:    6   .56250       18.84725
  Gamma Profile:    7   .57500       18.84734
  Gamma Profile:    8   .58750       18.84717
  Gamma Profile:    9   .60000       18.84847
  Gamma Profile:   10   .61250       18.84722
  Gamma Profile:   11   .62500       18.84749
  Gamma Profile:   12   .63750       18.84717
```

```
 Gamma Profile:   13   .65000       18.84728
 Gamma Profile:   14   .66250       18.84740
 Gamma Profile:   15   .67500       18.84718
 Gamma Profile:   16   .68750       18.84719
 Gamma Profile:   17   .70000       18.84739
 Gamma Profile:   18   .71250       18.84721
 Gamma Profile:   19   .72500       18.84729
 Gamma Profile:   20   .73750       18.84718
 Gamma Profile:   21   .75000       18.84719
 Gamma Profile:   22   .76250       18.84724
 Gamma Profile:   23   .77500       18.84719
 Gamma Profile:   24   .78750       18.84719
 Gamma Profile:   25   .80000       18.84712
 Gamma Profile:   26   .81250       18.84717
 Gamma Profile:   27   .82500       18.84713
 Gamma Profile:   28   .83750       18.84717
 Gamma Profile:   29   .85000       18.84714
 Gamma Profile:   30   .86250       18.84716
 Gamma Profile:   31   .87500       18.84717
 Gamma Profile:   32   .88750       18.84717
 Gamma Profile:   33   .90000       18.84717
 Gamma Profile:   34   .91250       18.84716
 Gamma Profile:   35   .92500       18.84712
 Gamma Profile:   36   .93750       18.84717
 Gamma Profile:   37   .95000       18.84717
 Gamma Profile:   38   .96250       18.84719
 Gamma Profile:   39   .97500       18.84716
 Gamma Profile:   40   .98750       18.84714
 Gamma Profile:   41  1.00000       18.84715
Profile likelihood for Efficiency:
                 index   Effic   LogLikelihood
 Effic Profile:    1   .01000       22.06406
 Effic Profile:    2   .01000       22.06406
 Effic Profile:    3   .01000       22.06406
 Effic Profile:    4   .01000       22.06406
 Effic Profile:    5   .01000       22.06406
 Effic Profile:    6   .01000       22.06406
 Effic Profile:    7   .01000       22.06413
 Effic Profile:    8   .02466       21.88798
 Effic Profile:    9   .04916       21.58858
 Effic Profile:   10   .07366       21.28423
 Effic Profile:   11   .09816       20.97710
 Effic Profile:   12   .12266       20.67023
 Effic Profile:   13   .14716       20.36733
 Effic Profile:   14   .17166       20.07335
 Effic Profile:   15   .19616       19.79407
 Effic Profile:   16   .22066       19.53641
 Effic Profile:   17   .24516       19.30788
 Effic Profile:   18   .26966       19.11665
 Effic Profile:   19   .29416       18.97099
 Effic Profile:   20   .31866       18.87886
 Effic Profile:   21   .34316       18.84712
 Effic Profile:   22   .36766       18.88100
 Effic Profile:   23   .39216       18.98079
 Effic Profile:   24   .41666       19.14321
 Effic Profile:   25   .44116       19.35908
 Effic Profile:   26   .46566       19.61470
 Effic Profile:   27   .49016       19.89593
 Effic Profile:   28   .51466       20.19067
 Effic Profile:   29   .53916       20.48992
 Effic Profile:   30   .56366       20.78736
 Effic Profile:   31   .58816       21.07873
 Effic Profile:   32   .61266       21.36156
 Effic Profile:   33   .63716       21.63418
 Effic Profile:   34   .66166       21.89580
 Effic Profile:   35   .68616       22.14609
 Effic Profile:   36   .71066       22.38502
 Effic Profile:   37   .73516       22.61278
 Effic Profile:   38   .75966       22.82971
 Effic Profile:   39   .78416       23.03620
 Effic Profile:   40   .80866       23.23267
```

```
  Effic Profile:   41   .83316       23.41956

 Profile likelihood for Density:
                 index   Density  LogLikelihood
Density Profile:    1      .00004289    32.73495102
Density Profile:    2      .00004289    32.73495102
Density Profile:    3      .00004289    32.73495102
Density Profile:    4      .00004289    32.73495102
Density Profile:    5      .00004289    32.73495102
Density Profile:    6      .00004289    32.73495102
Density Profile:    7      .00004289    32.73495102
Density Profile:    8      .00004289    32.73495102
Density Profile:    9      .00004289    32.73495102
Density Profile:   10      .00004289    32.73495102
Density Profile:   11      .00004289    32.73495102
Density Profile:   12      .00004783    31.79147550
Density Profile:   13      .00006575    29.36791727
Density Profile:   14      .00008368    27.36068118
Density Profile:   15      .00010160    25.53776767
Density Profile:   16      .00011953    23.80610519
Density Profile:   17      .00013745    22.14996151
Density Profile:   18      .00015538    20.66477128
Density Profile:   19      .00017330    19.56606110
Density Profile:   20      .00019122    18.99376927
Density Profile:   21      .00020915    18.84712454
Density Profile:   22      .00022707    18.93911522
Density Profile:   23      .00024500    19.13170480
Density Profile:   24      .00026292    19.35074839
Density Profile:   25      .00028085    19.56341772
Density Profile:   26      .00029877    19.75789000
Density Profile:   27      .00031670    19.93152506
Density Profile:   28      .00033462    20.08523770
Density Profile:   29      .00035254    20.22122056
Density Profile:   30      .00037047    20.34177710
Density Profile:   31      .00038839    20.44904864
Density Profile:   32      .00040632    20.54497397
Density Profile:   33      .00042424    20.63108459
Density Profile:   34      .00044217    20.70878269
Density Profile:   35      .00046009    20.78082648
Density Profile:   36      .00047802    20.84332867
Density Profile:   37      .00049594    20.90188410
Density Profile:   38      .00051386    20.95555728
Density Profile:   39      .00053179    21.00498311
Density Profile:   40      .00054971    21.05056544
Density Profile:   41      .00056764    21.09278604
```

Experiment #2

F/V Mary K Depletion Exp#2- Monkfish Cooperative Survey 2009 Cookie sweep

```
 Initial Values of parameters
    .00009                    =Mean  density of monkfish per sq ft
    .60000                    =Efficiency of trawl
 800.00000                    =K parameter for negative binomial dist
    .79900                    =Gamma parameter;  initial guess=trawl width/cell width
 Bounds on parameters
  Param #             Lower Bound                Upper Bound
          1   1.000000000000000E-007   2.000000000000000E-002
          2   5.000000000000000E-002   9.500000000000000E-001
          3   5.000000000000000E-001     2000.100000000000000
          4   8.000000000000000E-001   8.001000000000000E-001

 max # tows=          3
 Max number of hits =          3

 Value of likelihood function at initial guess:
     .000090     .600000  800.000000     .799000       12.9583367180

  Starting Pt    Density  Efficiency      K Parameter   Gamma Par  Likelihood Fc
 n
       Init Cond     .000049     .949991 1862.923555     .800029        7.189986
 Restart from IC     .000049     .950000 1998.328785     .800100        7.189101
 At 0.75 Current     .000049     .950000 1999.483338     .800022        7.189511
 At 1.25 Current     .000049     .950000 2000.100000     .800093        7.189177
      At Current     .000049     .950000 2000.100000     .800100        7.189142
 At 1.25 IniCond     .000532     .050000     .500000     .800082       11.940938
      At Current     .000044     .950000     .500000     .800040       11.618120
 At 0.75 IniCond     .000049     .949999 1991.254583     .800001        7.189656
      At Current     .000049     .949999 1991.254583     .800001        7.189656

BEST soln=      .00005      .95000  1998.32878      .80010    7.1891009854
   Ave Density/ft^2        .0000492
   Efficiency              .9499999
   K Parameter           1998.32878
   Gamma Parameter           .80010

Profile range for m=         .0000361333         .0000650994
Profile range for e=         .7270126202        1.4798501708
Profile range for k=  -187550.6667816964   -33727.3512383339
Profile range for g=         .6117690540        1.1238158320

 Profile likelihood for Gamma:
                 index   gamma   LogLikelihood
  Gamma Profile:    1   .50000        9.98117
  Gamma Profile:    2   .51250        9.82899
  Gamma Profile:    3   .52500        9.67919
  Gamma Profile:    4   .53750        9.53196
  Gamma Profile:    5   .55000        9.38727
  Gamma Profile:    6   .56250        9.24524
  Gamma Profile:    7   .57500        9.10595
  Gamma Profile:    8   .58750        8.96947
  Gamma Profile:    9   .60000        8.83598
  Gamma Profile:   10   .61250        8.70544
  Gamma Profile:   11   .62500        8.57807
  Gamma Profile:   12   .63750        8.45390
  Gamma Profile:   13   .65000        8.33306
  Gamma Profile:   14   .66250        8.21563
  Gamma Profile:   15   .67500        8.10179
  Gamma Profile:   16   .68750        7.99161
  Gamma Profile:   17   .70000        7.88524
  Gamma Profile:   18   .71250        7.78282
  Gamma Profile:   19   .72500        7.68448
  Gamma Profile:   20   .73750        7.59040
  Gamma Profile:   21   .75000        7.50070
  Gamma Profile:   22   .76250        7.41560
  Gamma Profile:   23   .77500        7.33525
  Gamma Profile:   24   .78750        7.25987
  Gamma Profile:   25   .80000        7.18964
  Gamma Profile:   26   .81250        7.12483
```

```
  Gamma Profile:   27   .82500        7.06565
  Gamma Profile:   28   .83750        7.01238
  Gamma Profile:   29   .85000        6.96531
  Gamma Profile:   30   .86250        6.92474
  Gamma Profile:   31   .87500        6.89103
  Gamma Profile:   32   .88750        6.86454
  Gamma Profile:   33   .90000        6.84568
  Gamma Profile:   34   .91250        6.83491
  Gamma Profile:   35   .92500        6.83271
  Gamma Profile:   36   .93750        6.83966
  Gamma Profile:   37   .95000        6.85636
  Gamma Profile:   38   .96250        6.88351
  Gamma Profile:   39   .97500        6.92186
  Gamma Profile:   40   .98750        6.97229
  Gamma Profile:   41  1.00000        7.03577
 Profile likelihood for Efficiency:
                 index   Effic   LogLikelihood
  Effic Profile:    1   .46000        9.61531
  Effic Profile:    2   .48450        9.53437
  Effic Profile:    3   .50900        9.45042
  Effic Profile:    4   .53350        9.36327
  Effic Profile:    5   .55800        9.27268
  Effic Profile:    6   .58250        9.17839
  Effic Profile:    7   .60700        9.08012
  Effic Profile:    8   .63150        8.97750
  Effic Profile:    9   .65600        8.87013
  Effic Profile:   10   .68050        8.75750
  Effic Profile:   11   .70500        8.63897
  Effic Profile:   12   .72950        8.51370
  Effic Profile:   13   .75400        8.38056
  Effic Profile:   14   .77850        8.23781
  Effic Profile:   15   .80300        8.08957
  Effic Profile:   16   .82750        7.91119
  Effic Profile:   17   .85200        7.74270
  Effic Profile:   18   .87650        7.58525
  Effic Profile:   19   .90100        7.44008
  Effic Profile:   20   .92550        7.30772
  Effic Profile:   21   .95000        7.18909
  Effic Profile:   22   .97450        7.08519
  Effic Profile:   23   .99900        6.99714
  Effic Profile:   24  1.00000        6.99393
  Effic Profile:   25  1.00000        6.99393
  Effic Profile:   26  1.00000        6.99393
  Effic Profile:   27  1.00000        6.99393
  Effic Profile:   28  1.00000        6.99393
  Effic Profile:   29  1.00000        6.99393
  Effic Profile:   30  1.00000        6.99393
  Effic Profile:   31  1.00000        6.99393
  Effic Profile:   32  1.00000        6.99393
  Effic Profile:   33  1.00000        6.99393
  Effic Profile:   34  1.00000        6.99393
  Effic Profile:   35  1.00000        6.99393
  Effic Profile:   36  1.00000        6.99393
  Effic Profile:   37  1.00000        6.99393
  Effic Profile:   38  1.00000        6.99393
  Effic Profile:   39  1.00000        6.99393
  Effic Profile:   40  1.00000        6.99393
  Effic Profile:   41  1.00000        6.99393

 Profile likelihood for Density:
                 index   Density  LogLikelihood
Density Profile:    1      .00000903    14.67708362
Density Profile:    2      .00000903    14.67708362
Density Profile:    3      .00000903    14.67708362
Density Profile:    4      .00000903    14.67708362
Density Profile:    5      .00000903    14.67708362
Density Profile:    6      .00000903    14.67708362
Density Profile:    7      .00000903    14.67708362
Density Profile:    8      .00000903    14.67708362
Density Profile:    9      .00000903    14.67708362
Density Profile:   10      .00000903    14.67708362
```

Density Profile:	11	.00000903	14.67708362
Density Profile:	12	.00000903	14.67708362
Density Profile:	13	.00001193	13.51857810
Density Profile:	14	.00001659	12.26187423
Density Profile:	15	.00002124	11.21068082
Density Profile:	16	.00002590	10.27035365
Density Profile:	17	.00003056	9.41030648
Density Profile:	18	.00003521	8.63382913
Density Profile:	19	.00003987	7.95897367
Density Profile:	20	.00004453	7.40291620
Density Profile:	21	.00004918	7.18909808
Density Profile:	22	.00005384	7.37641876
Density Profile:	23	.00005850	7.89757678
Density Profile:	24	.00006315	8.65841376
Density Profile:	25	.00006781	9.20058434
Density Profile:	26	.00007247	9.48268241
Density Profile:	27	.00007712	9.64930800
Density Profile:	28	.00008178	9.76904606
Density Profile:	29	.00008644	9.86166675
Density Profile:	30	.00009109	9.93638072
Density Profile:	31	.00009575	9.99836245
Density Profile:	32	.00010041	10.05082710
Density Profile:	33	.00010506	10.09593841
Density Profile:	34	.00010972	10.13524053
Density Profile:	35	.00011438	10.16981990
Density Profile:	36	.00011903	10.20051734
Density Profile:	37	.00012369	10.22798166
Density Profile:	38	.00012835	10.25269955
Density Profile:	39	.00013300	10.27509767
Density Profile:	40	.00013766	10.29547975
Density Profile:	41	.00014232	10.31411393

F/V Mary K Depletion Exp#3- Monkfish Cooperative Survey 2009 Cookie sweep

```
 Initial Values of parameters
    .00009                    =Mean  density of monkfish per sq ft
    .60000                    =Efficiency of trawl
 800.00000                    =K parameter for negative binomial dist
    .79900                    =Gamma parameter;  initial guess=trawl width/cell width
 Bounds on parameters
  Param #            Lower Bound                Upper Bound
          1   1.000000000000000E-007   2.000000000000000E-002
          2   5.000000000000000E-002   9.500000000000000E-001
          3   5.000000000000000E-001     2000.100000000000000
          4   8.000000000000000E-001   8.001000000000000E-001

 max # tows=          5
 Max number of hits =          5

 Value of likelihood function at initial guess:
     .000090     .600000  800.000000     .799000       14.0994794028

  Starting Pt    Density  Efficiency      K Parameter   Gamma Par  Likelihood Fc
 n
       Init Cond     .000092     .545448   36.092252     .800091       13.944755
 Restart from IC     .000092     .545448   36.092252     .800091       13.944755
 At 0.75 Current     .000092     .545530   35.800788     .800049       13.944757
 At 1.25 Current     .000092     .545382   35.812397     .800099       13.944759
      At Current     .000092     .545382   35.812397     .800099       13.944759
 At 1.25 IniCond     .000092     .545614   35.949027     .800100       13.944763
      At Current     .000092     .545614   35.949027     .800100       13.944763
 At 0.75 IniCond     .000092     .545762   36.242183     .800000       13.944825
      At Current     .000092     .545569   35.957433     .800000       13.944765

BEST soln=      .00009      .54545    36.09225      .80009   13.9447546404
   Ave Density/ft^2        .0000921
   Efficiency              .5454478
   K Parameter             36.09225
   Gamma Parameter           .80009

Profile range for m=         .0000692495         .0001213348
Profile range for e=         .3677967841         .7499060133
Profile range for k=    -6181.7413638744      -99.1528281636
Profile range for g=         .5803031893        1.0210431831

 Profile likelihood for Gamma:
                index   gamma   LogLikelihood
  Gamma Profile:    1   .50000       13.94477
  Gamma Profile:    2   .51250       13.96250
  Gamma Profile:    3   .52500       13.96113
  Gamma Profile:    4   .53750       13.96674
  Gamma Profile:    5   .55000       13.96781
  Gamma Profile:    6   .56250       13.96107
  Gamma Profile:    7   .57500       13.96278
  Gamma Profile:    8   .58750       13.95899
  Gamma Profile:    9   .60000       13.95870
  Gamma Profile:   10   .61250       13.95840
  Gamma Profile:   11   .62500       13.95581
  Gamma Profile:   12   .63750       13.95407
  Gamma Profile:   13   .65000       13.94473
  Gamma Profile:   14   .66250       13.95276
  Gamma Profile:   15   .67500       13.94974
  Gamma Profile:   16   .68750       13.95028
  Gamma Profile:   17   .70000       13.94938
  Gamma Profile:   18   .71250       13.94819
  Gamma Profile:   19   .72500       13.94737
  Gamma Profile:   20   .73750       13.94673
  Gamma Profile:   21   .75000       13.94599
  Gamma Profile:   22   .76250       13.94548
  Gamma Profile:   23   .77500       13.94512
  Gamma Profile:   24   .78750       13.94478
  Gamma Profile:   25   .80000       13.94476
  Gamma Profile:   26   .81250       13.94476
  Gamma Profile:   27   .82500       13.94477
```

```
  Gamma Profile:   28   .83750       13.94474
  Gamma Profile:   29   .85000       13.94478
  Gamma Profile:   30   .86250       13.94477
  Gamma Profile:   31   .87500       13.94476
  Gamma Profile:   32   .88750       13.94476
  Gamma Profile:   33   .90000       13.94475
  Gamma Profile:   34   .91250       13.94477
  Gamma Profile:   35   .92500       13.94476
  Gamma Profile:   36   .93750       13.94476
  Gamma Profile:   37   .95000       14.06950
  Gamma Profile:   38   .96250       14.06820
  Gamma Profile:   39   .97500       14.05346
  Gamma Profile:   40   .98750       14.06817
  Gamma Profile:   41  1.00000       13.94476
 Profile likelihood for Efficiency:
                 index   Effic   LogLikelihood
  Effic Profile:    1   .05545       17.42447
  Effic Profile:    2   .07995       17.24145
  Effic Profile:    3   .10445       17.05217
  Effic Profile:    4   .12895       16.85664
  Effic Profile:    5   .15345       16.65516
  Effic Profile:    6   .17795       16.44815
  Effic Profile:    7   .20245       16.23624
  Effic Profile:    8   .22695       16.02039
  Effic Profile:    9   .25145       15.80185
  Effic Profile:   10   .27595       15.58227
  Effic Profile:   11   .30045       15.36369
  Effic Profile:   12   .32495       15.14857
  Effic Profile:   13   .34945       14.93980
  Effic Profile:   14   .37395       14.74065
  Effic Profile:   15   .39845       14.55473
  Effic Profile:   16   .42295       14.38596
  Effic Profile:   17   .44745       14.23837
  Effic Profile:   18   .47195       14.11620
  Effic Profile:   19   .49645       14.02374
  Effic Profile:   20   .52095       13.96518
  Effic Profile:   21   .54545       13.94475
  Effic Profile:   22   .56995       13.96649
  Effic Profile:   23   .59445       14.03433
  Effic Profile:   24   .61895       14.15199
  Effic Profile:   25   .64345       14.32279
  Effic Profile:   26   .66795       14.54890
  Effic Profile:   27   .69245       14.84101
  Effic Profile:   28   .71695       15.20452
  Effic Profile:   29   .74145       15.63987
  Effic Profile:   30   .76595       16.12482
  Effic Profile:   31   .79045       16.59080
  Effic Profile:   32   .81495       17.04521
  Effic Profile:   33   .83945       17.48475
  Effic Profile:   34   .86395       17.90852
  Effic Profile:   35   .88845       18.31713
  Effic Profile:   36   .91295       18.71207
  Effic Profile:   37   .93745       19.09527
  Effic Profile:   38   .96195       19.46891
  Effic Profile:   39   .98645       19.83540
  Effic Profile:   40  1.00000       20.03604
  Effic Profile:   41  1.00000       20.03604

 Profile likelihood for Density:
                 index   Density  LogLikelihood
Density Profile:    1      .00001731    25.56181150
Density Profile:    2      .00001731    25.56181150
Density Profile:    3      .00001731    25.56181150
Density Profile:    4      .00001731    25.56181150
Density Profile:    5      .00001731    25.56181150
Density Profile:    6      .00001731    25.56181150
Density Profile:    7      .00001731    25.56181150
Density Profile:    8      .00001731    25.56181150
Density Profile:    9      .00001731    25.56181150
Density Profile:   10      .00001731    25.56181150
Density Profile:   11      .00001731    25.56181151
```

Density Profile:	12	.00001731	25.56181151
Density Profile:	13	.00002273	23.47801918
Density Profile:	14	.00003140	21.46240278
Density Profile:	15	.00004007	19.80730625
Density Profile:	16	.00004874	18.33316968
Density Profile:	17	.00005740	16.98222693
Density Profile:	18	.00006607	15.77141495
Density Profile:	19	.00007474	14.78670111
Density Profile:	20	.00008341	14.15102425
Density Profile:	21	.00009207	13.94475464
Density Profile:	22	.00010074	14.10296967
Density Profile:	23	.00010941	14.46286550
Density Profile:	24	.00011807	14.88937456
Density Profile:	25	.00012674	15.29159145
Density Profile:	26	.00013541	15.61675777
Density Profile:	27	.00014408	15.86659576
Density Profile:	28	.00015274	16.06204295
Density Profile:	29	.00016141	16.21935583
Density Profile:	30	.00017008	16.34908008
Density Profile:	31	.00017875	16.45819755
Density Profile:	32	.00018741	16.55143156
Density Profile:	33	.00019608	16.63215148
Density Profile:	34	.00020475	16.70278845
Density Profile:	35	.00021342	16.76518950
Density Profile:	36	.00022208	16.82076681
Density Profile:	37	.00023075	16.87059240
Density Profile:	38	.00023942	16.91553998
Density Profile:	39	.00024808	16.95631280
Density Profile:	40	.00025675	16.99511030
Density Profile:	41	.00026542	17.03349548

F/V Mary K Depletion Exp#4- Monkfish Cooperative Survey 2009 Cookie sweep

```
 Initial Values of parameters
    .00009                   =Mean  density of monkfish per sq ft
    .60000                   =Efficiency of trawl
 800.00000                   =K parameter for negative binomial dist
    .79900                   =Gamma parameter;  initial guess=trawl width/cell width
 Bounds on parameters
  Param #            Lower Bound                Upper Bound
          1   1.000000000000000E-007   2.000000000000000E-002
          2   5.000000000000000E-002   9.500000000000000E-001
          3   5.000000000000000E-001     2000.100000000000000
          4   8.000000000000000E-001   8.001000000000000E-001

 max # tows=          5
 Max number of hits =          5

 Value of likelihood function at initial guess:
     .000090     .600000  800.000000     .799000       16.5239544254

  Starting Pt    Density  Efficiency      K Parameter   Gamma Par  Likelihood Fc
 n
       Init Cond     .000102     .682002  868.548230     .800001       14.863495
 Restart from IC     .000102     .682016 1981.628861     .800000       14.861226
 At 0.75 Current     .000102     .682277 1662.014562     .800011       14.861580
 At 1.25 Current     .000102     .682246 1704.920199     .800099       14.861468
      At Current     .000102     .682246 1704.920199     .800099       14.861468
 At 1.25 IniCond     .000102     .682832  921.197851     .800100       14.863254
      At Current     .000102     .682239 1992.557789     .800099       14.861188
 At 0.75 IniCond     .000098     .653701     .500000     .800100       20.203233
      At Current     .000098     .653701     .500000     .800100       20.203233

BEST soln=      .00010      .68224  1992.55779      .80010   14.8611882004
   Ave Density/ft^2        .0001022
   Efficiency              .6822388
   K Parameter           1992.55779
   Gamma Parameter           .80010

Profile range for m=         .0000827484         .0001243961
Profile range for e=         .5255430255         .8457045331
Profile range for k= -3486279.5321977080   -60759.9211378478
Profile range for g=         .6471605847         .9582943972

 Profile likelihood for Gamma:
                index   gamma   LogLikelihood
  Gamma Profile:    1   .50000       15.34617
  Gamma Profile:    2   .51250       15.19854
  Gamma Profile:    3   .52500       15.07721
  Gamma Profile:    4   .53750       14.98244
  Gamma Profile:    5   .55000       14.91467
  Gamma Profile:    6   .56250       14.87414
  Gamma Profile:    7   .57500       14.86122
  Gamma Profile:    8   .58750       14.86120
  Gamma Profile:    9   .60000       14.86126
  Gamma Profile:   10   .61250       14.86124
  Gamma Profile:   11   .62500       14.86122
  Gamma Profile:   12   .63750       14.86124
  Gamma Profile:   13   .65000       14.86123
  Gamma Profile:   14   .66250       14.86122
  Gamma Profile:   15   .67500       14.86122
  Gamma Profile:   16   .68750       14.86124
  Gamma Profile:   17   .70000       14.86120
  Gamma Profile:   18   .71250       14.86118
  Gamma Profile:   19   .72500       14.86120
  Gamma Profile:   20   .73750       14.86120
  Gamma Profile:   21   .75000       14.86121
  Gamma Profile:   22   .76250       14.86121
  Gamma Profile:   23   .77500       14.86123
  Gamma Profile:   24   .78750       14.86122
  Gamma Profile:   25   .80000       14.86119
  Gamma Profile:   26   .81250       14.86122
  Gamma Profile:   27   .82500       14.86120
```

```
  Gamma Profile:   28   .83750       14.86120
  Gamma Profile:   29   .85000       14.86121
  Gamma Profile:   30   .86250       14.86122
  Gamma Profile:   31   .87500       14.86124
  Gamma Profile:   32   .88750       14.86124
  Gamma Profile:   33   .90000       14.86123
  Gamma Profile:   34   .91250       14.86125
  Gamma Profile:   35   .92500       14.86121
  Gamma Profile:   36   .93750       14.86120
  Gamma Profile:   37   .95000       14.86122
  Gamma Profile:   38   .96250       14.86122
  Gamma Profile:   39   .97500       14.86122
  Gamma Profile:   40   .98750       14.86124
  Gamma Profile:   41  1.00000       14.86120
 Profile likelihood for Efficiency:
                 index   Effic   LogLikelihood
  Effic Profile:    1   .19224       18.35356
  Effic Profile:    2   .21674       18.19560
  Effic Profile:    3   .24124       18.03269
  Effic Profile:    4   .26574       17.86483
  Effic Profile:    5   .29024       17.69195
  Effic Profile:    6   .31474       17.51414
  Effic Profile:    7   .33924       17.33153
  Effic Profile:    8   .36374       17.14430
  Effic Profile:    9   .38824       16.95286
  Effic Profile:   10   .41274       16.75761
  Effic Profile:   11   .43724       16.55930
  Effic Profile:   12   .46174       16.35874
  Effic Profile:   13   .48624       16.15703
  Effic Profile:   14   .51074       15.95560
  Effic Profile:   15   .53524       15.75611
  Effic Profile:   16   .55974       15.56042
  Effic Profile:   17   .58424       15.45261
  Effic Profile:   18   .60874       15.19643
  Effic Profile:   19   .63324       15.01121
  Effic Profile:   20   .65774       14.89889
  Effic Profile:   21   .68224       14.86119
  Effic Profile:   22   .70674       14.89962
  Effic Profile:   23   .73124       15.01523
  Effic Profile:   24   .75574       15.20930
  Effic Profile:   25   .78024       15.48290
  Effic Profile:   26   .80474       15.83695
  Effic Profile:   27   .82924       16.27203
  Effic Profile:   28   .85374       16.78852
  Effic Profile:   29   .87824       17.38679
  Effic Profile:   30   .90274       18.06661
  Effic Profile:   31   .92724       18.82760
  Effic Profile:   32   .95174       19.66895
  Effic Profile:   33   .97624       20.58926
  Effic Profile:   34  1.00000       21.55651
  Effic Profile:   35  1.00000       21.55651
  Effic Profile:   36  1.00000       21.55651
  Effic Profile:   37  1.00000       21.55651
  Effic Profile:   38  1.00000       21.55651
  Effic Profile:   39  1.00000       21.55651
  Effic Profile:   40  1.00000       21.55651
  Effic Profile:   41  1.00000       21.55651

 Profile likelihood for Density:
                 index   Density  LogLikelihood
Density Profile:    1      .00002069    25.01742185
Density Profile:    2      .00002069    25.01742185
Density Profile:    3      .00002069    25.01742185
Density Profile:    4      .00002069    25.01742185
Density Profile:    5      .00002069    25.01742185
Density Profile:    6      .00002069    25.01742185
Density Profile:    7      .00002069    25.01742185
Density Profile:    8      .00002069    25.01742185
Density Profile:    9      .00002069    25.01742185
Density Profile:   10      .00002069    25.01742185
Density Profile:   11      .00002069    25.01742185
```

Density Profile:	12	.00002284	24.28149099
Density Profile:	13	.00003165	22.27642647
Density Profile:	14	.00004046	20.75743609
Density Profile:	15	.00004928	19.46661618
Density Profile:	16	.00005809	18.33493713
Density Profile:	17	.00006690	17.33333182
Density Profile:	18	.00007571	16.45626096
Density Profile:	19	.00008453	15.71509755
Density Profile:	20	.00009334	15.18135167
Density Profile:	21	.00010215	14.86118820
Density Profile:	22	.00011096	15.13632710
Density Profile:	23	.00011978	15.87662831
Density Profile:	24	.00012859	16.26990592
Density Profile:	25	.00013740	16.73038242
Density Profile:	26	.00014621	17.12874931
Density Profile:	27	.00015503	17.47482022
Density Profile:	28	.00016384	17.77746554
Density Profile:	29	.00017265	18.04419522
Density Profile:	30	.00018146	18.28107844
Density Profile:	31	.00019028	18.49298675
Density Profile:	32	.00019909	18.68364905
Density Profile:	33	.00020790	18.85609296
Density Profile:	34	.00021671	19.01403498
Density Profile:	35	.00022553	19.16436738
Density Profile:	36	.00023434	19.30793532
Density Profile:	37	.00024315	19.44496702
Density Profile:	38	.00025196	19.57578981
Density Profile:	39	.00026078	19.70074728
Density Profile:	40	.00026959	19.82011445
Density Profile:	41	.00027840	19.93430026

F/V Endurance Depletion Exp#5- Monkfish Cooperative Survey 2009 Cookie sweep

```
 Initial Values of parameters
    .00009              =Mean  density of monkfish per sq ft
    .60000              =Efficiency of trawl
 800.00000              =K parameter for negative binomial dist
    .79900              =Gamma parameter;  initial guess=trawl width/cell width
 Bounds on parameters
  Param #            Lower Bound                Upper Bound
          1   1.000000000000000E-007   2.000000000000000E-002
          2   5.000000000000000E-002   9.500000000000000E-001
          3   5.000000000000000E-001     2000.100000000000000
          4   8.000000000000000E-001   8.001000000000000E-001

 max # tows=          6
 Max number of hits =          6

 Value of likelihood function at initial guess:
     .000090     .600000  800.000000     .799000       40.9830258445

  Starting Pt    Density  Efficiency      K Parameter   Gamma Par  Likelihood Fc
 n
       Init Cond     .000170     .429534     .500000     .800096       26.553342
 Restart from IC     .000170     .429403     .500000     .800096       26.553342
 At 0.75 Current     .000174     .382392   30.409138     .800006       20.015653
 At 1.25 Current     .000174     .382461   30.604524     .800079       20.015675
      At Current     .000174     .382329   30.759826     .800079       20.015641
 At 1.25 IniCond     .000752     .050000     .500000     .800094       26.846228
      At Current     .000752     .050000     .500000     .800096       26.846228
 At 0.75 IniCond     .000170     .429437     .500000     .800100       26.553342
      At Current     .000170     .429437     .500000     .800100       26.553342

BEST soln=      .00017      .38233    30.75983      .80008   20.0156405995
   Ave Density/ft^2        .0001737
   Efficiency              .3823286
   K Parameter             30.75983
   Gamma Parameter           .80008

Profile range for m=         .0001384769         .0002175830
Profile range for e=         .2647723973         .5501045631
Profile range for k=    -4221.3216735437      -68.4412971559
Profile range for g=         .5861721632        1.0145777592

 Profile likelihood for Gamma:
                index   gamma   LogLikelihood
  Gamma Profile:    1   .50000       20.01777
  Gamma Profile:    2   .51250       20.03655
  Gamma Profile:    3   .52500       20.02794
  Gamma Profile:    4   .53750       20.01905
  Gamma Profile:    5   .55000       20.01565
  Gamma Profile:    6   .56250       20.01567
  Gamma Profile:    7   .57500       20.02812
  Gamma Profile:    8   .58750       20.03666
  Gamma Profile:    9   .60000       20.01565
  Gamma Profile:   10   .61250       20.01573
  Gamma Profile:   11   .62500       20.01569
  Gamma Profile:   12   .63750       20.01928
  Gamma Profile:   13   .65000       20.01817
  Gamma Profile:   14   .66250       20.01576
  Gamma Profile:   15   .67500       20.01568
  Gamma Profile:   16   .68750       20.01565
  Gamma Profile:   17   .70000       20.01566
  Gamma Profile:   18   .71250       20.01566
  Gamma Profile:   19   .72500       20.01563
  Gamma Profile:   20   .73750       20.01567
  Gamma Profile:   21   .75000       20.01566
  Gamma Profile:   22   .76250       20.01564
  Gamma Profile:   23   .77500       20.01567
  Gamma Profile:   24   .78750       20.01570
  Gamma Profile:   25   .80000       20.01564
  Gamma Profile:   26   .81250       20.01564
  Gamma Profile:   27   .82500       20.01566
```

```
 Gamma Profile:   28   .83750       20.01567
 Gamma Profile:   29   .85000       20.01564
 Gamma Profile:   30   .86250       20.01566
 Gamma Profile:   31   .87500       20.01567
 Gamma Profile:   32   .88750       20.01566
 Gamma Profile:   33   .90000       20.01563
 Gamma Profile:   34   .91250       20.01568
 Gamma Profile:   35   .92500       20.01566
 Gamma Profile:   36   .93750       20.01566
 Gamma Profile:   37   .95000       20.01567
 Gamma Profile:   38   .96250       20.01565
 Gamma Profile:   39   .97500       20.01567
 Gamma Profile:   40   .98750       20.01569
 Gamma Profile:   41  1.00000       20.01566
Profile likelihood for Efficiency:
                 index   Effic   LogLikelihood
 Effic Profile:    1   .01000       23.60693
 Effic Profile:    2   .01000       23.60693
 Effic Profile:    3   .01000       23.60693
 Effic Profile:    4   .01000       23.60693
 Effic Profile:    5   .01000       23.60693
 Effic Profile:    6   .01483       23.56199
 Effic Profile:    7   .03933       23.32904
 Effic Profile:    8   .06383       23.08756
 Effic Profile:    9   .08833       22.83729
 Effic Profile:   10   .11283       22.57814
 Effic Profile:   11   .13733       22.31008
 Effic Profile:   12   .16183       22.03347
 Effic Profile:   13   .18633       21.74923
 Effic Profile:   14   .21083       21.45908
 Effic Profile:   15   .23533       21.16659
 Effic Profile:   16   .25983       20.87809
 Effic Profile:   17   .28433       20.60447
 Effic Profile:   18   .30883       20.36254
 Effic Profile:   19   .33333       20.17315
 Effic Profile:   20   .35783       20.05490
 Effic Profile:   21   .38233       20.01564
 Effic Profile:   22   .40683       20.05030
 Effic Profile:   23   .43133       20.14553
 Effic Profile:   24   .45583       20.28647
 Effic Profile:   25   .48033       20.46015
 Effic Profile:   26   .50483       20.65624
 Effic Profile:   27   .52933       20.86695
 Effic Profile:   28   .55383       21.08641
 Effic Profile:   29   .57833       21.31011
 Effic Profile:   30   .60283       21.53481
 Effic Profile:   31   .62733       21.75799
 Effic Profile:   32   .65183       21.97782
 Effic Profile:   33   .67633       22.19299
 Effic Profile:   34   .70083       22.40280
 Effic Profile:   35   .72533       22.60661
 Effic Profile:   36   .74983       22.80418
 Effic Profile:   37   .77433       22.99558
 Effic Profile:   38   .79883       23.18090
 Effic Profile:   39   .82333       23.36042
 Effic Profile:   40   .84783       23.53441
 Effic Profile:   41   .87233       26.88127

Profile likelihood for Density:
                 index   Density  LogLikelihood
Density Profile:    1      .00003462    32.92238134
Density Profile:    2      .00003462    32.92238134
Density Profile:    3      .00003462    32.92238134
Density Profile:    4      .00003462    32.92238134
Density Profile:    5      .00003462    32.92238134
Density Profile:    6      .00003462    32.92238134
Density Profile:    7      .00003462    32.92238134
Density Profile:    8      .00003462    32.92238134
Density Profile:    9      .00003462    32.92238134
Density Profile:   10      .00003462    32.92238134
Density Profile:   11      .00003462    32.92238140
```

Density Profile:	12	.00003466	32.91023408
Density Profile:	13	.00005011	29.83649894
Density Profile:	14	.00006557	27.79873024
Density Profile:	15	.00008102	26.08502144
Density Profile:	16	.00009647	24.56107926
Density Profile:	17	.00011193	23.18231048
Density Profile:	18	.00012738	21.95273184
Density Profile:	19	.00014283	20.93142824
Density Profile:	20	.00015829	20.24507404
Density Profile:	21	.00017374	20.01564060
Density Profile:	22	.00018919	20.19415482
Density Profile:	23	.00020465	20.58213623
Density Profile:	24	.00022010	20.98920048
Density Profile:	25	.00023555	21.32803782
Density Profile:	26	.00025101	21.59403664
Density Profile:	27	.00026646	21.80434246
Density Profile:	28	.00028191	21.97447841
Density Profile:	29	.00029737	22.11497305
Density Profile:	30	.00031282	22.23327327
Density Profile:	31	.00032827	22.33435793
Density Profile:	32	.00034372	22.42182813
Density Profile:	33	.00035918	22.49837958
Density Profile:	34	.00037463	22.56600908
Density Profile:	35	.00039008	22.62617920
Density Profile:	36	.00040554	22.68011009
Density Profile:	37	.00042099	22.72874896
Density Profile:	38	.00043644	22.77289119
Density Profile:	39	.00045190	22.81307395
Density Profile:	40	.00046735	22.84986412
Density Profile:	41	.00048280	22.88368105

F/V Endurance Depletion Exp#6- Monkfish Cooperative Survey 2009 Roller sweep

```
 Initial Values of parameters
    .00009                        =Mean  density of monkfish per sq ft
    .60000                        =Efficiency of trawl
 800.00000                        =K parameter for negative binomial dist
    .79900                        =Gamma parameter;  initial guess=trawl width/cell width
 Bounds on parameters
  Param #             Lower Bound                Upper Bound
          1   1.000000000000000E-007   2.000000000000000E-002
          2   5.000000000000000E-002   9.500000000000000E-001
          3   5.000000000000000E-001     2000.100000000000000
          4   8.000000000000000E-001   8.001000000000000E-001

 max # tows=          6
 Max number of hits =          6

 Value of likelihood function at initial guess:
     .000090     .600000  800.000000     .799000      173.3437473240

  Starting Pt    Density  Efficiency      K Parameter   Gamma Par  Likelihood Fc
 n
       Init Cond     .000817     .050000     .500000     .800000       30.895870
 Restart from IC     .000817     .050000     .500000     .800000       30.895870
 At 0.75 Current     .000816     .050000     .500000     .800032       30.895871
 At 1.25 Current     .000811     .050000   17.641123     .800002       23.285576
      At Current     .000811     .050000   17.641123     .800002       23.285576
 At 1.25 IniCond     .000816     .050000     .500000     .800096       30.895873
      At Current     .000816     .050000     .500000     .800096       30.895873
 At 0.75 IniCond     .000816     .050000     .500000     .800000       30.895870
      At Current     .000816     .050000     .500000     .800000       30.895870

BEST soln=      .00081      .05000    17.64112      .80000   23.2855757860
   Ave Density/ft^2        .0008114
   Efficiency              .0500004
   K Parameter             17.64112
   Gamma Parameter           .80000

Profile range for m=         .0006480096         .0010221376
Profile range for e=         .0387413616         .0640136992
Profile range for k=       -6.4127423076      144.3646782037
Profile range for g=        -.6363943855        1.8487032289

 Profile likelihood for Gamma:
                index   gamma   LogLikelihood
  Gamma Profile:    1   .50000       23.06111
  Gamma Profile:    2   .51250       23.06966
  Gamma Profile:    3   .52500       23.07817
  Gamma Profile:    4   .53750       23.08683
  Gamma Profile:    5   .55000       23.09555
  Gamma Profile:    6   .56250       23.10431
  Gamma Profile:    7   .57500       23.11314
  Gamma Profile:    8   .58750       23.12210
  Gamma Profile:    9   .60000       23.13116
  Gamma Profile:   10   .61250       23.14022
  Gamma Profile:   11   .62500       23.14942
  Gamma Profile:   12   .63750       23.15868
  Gamma Profile:   13   .65000       23.16797
  Gamma Profile:   14   .66250       23.17781
  Gamma Profile:   15   .67500       23.18692
  Gamma Profile:   16   .68750       23.19649
  Gamma Profile:   17   .70000       23.20608
  Gamma Profile:   18   .71250       23.21578
  Gamma Profile:   19   .72500       23.22551
  Gamma Profile:   20   .73750       23.23529
  Gamma Profile:   21   .75000       23.24535
  Gamma Profile:   22   .76250       23.25518
  Gamma Profile:   23   .77500       23.26535
  Gamma Profile:   24   .78750       23.27556
  Gamma Profile:   25   .80000       23.28557
  Gamma Profile:   26   .81250       23.29587
  Gamma Profile:   27   .82500       23.30617
```

```
  Gamma Profile:   28   .83750       23.31675
  Gamma Profile:   29   .85000       23.32692
  Gamma Profile:   30   .86250       23.33761
  Gamma Profile:   31   .87500       23.34806
  Gamma Profile:   32   .88750       23.35870
  Gamma Profile:   33   .90000       23.36948
  Gamma Profile:   34   .91250       23.38019
  Gamma Profile:   35   .92500       23.39108
  Gamma Profile:   36   .93750       23.40201
  Gamma Profile:   37   .95000       23.41291
  Gamma Profile:   38   .96250       23.42390
  Gamma Profile:   39   .97500       23.43491
  Gamma Profile:   40   .98750       23.44624
  Gamma Profile:   41  1.00000       23.45737
 Profile likelihood for Efficiency:
                 index   Effic   LogLikelihood
  Effic Profile:    1   .01000       22.86302
  Effic Profile:    2   .01000       22.86302
  Effic Profile:    3   .01000       22.86302
  Effic Profile:    4   .01000       22.86302
  Effic Profile:    5   .01000       22.86302
  Effic Profile:    6   .01000       22.86302
  Effic Profile:    7   .01000       22.86302
  Effic Profile:    8   .01000       22.86302
  Effic Profile:    9   .01000       22.86302
  Effic Profile:   10   .01000       22.86302
  Effic Profile:   11   .01000       22.86302
  Effic Profile:   12   .01000       22.86302
  Effic Profile:   13   .01000       22.86302
  Effic Profile:   14   .01000       22.86302
  Effic Profile:   15   .01000       22.86302
  Effic Profile:   16   .01000       22.86302
  Effic Profile:   17   .01000       22.86302
  Effic Profile:   18   .01000       22.86302
  Effic Profile:   19   .01000       22.86302
  Effic Profile:   20   .02550       23.00131
  Effic Profile:   21   .05000       23.28557
  Effic Profile:   22   .07450       23.63625
  Effic Profile:   23   .09900       24.03563
  Effic Profile:   24   .12350       30.95560
  Effic Profile:   25   .14800       30.98451
  Effic Profile:   26   .17250       31.01823
  Effic Profile:   27   .19700       31.05693
  Effic Profile:   28   .22150       31.10079
  Effic Profile:   29   .24600       31.14997
  Effic Profile:   30   .27050       31.20463
  Effic Profile:   31   .29500       31.26491
  Effic Profile:   32   .31950       31.33092
  Effic Profile:   33   .34400       31.40279
  Effic Profile:   34   .36850       31.48057
  Effic Profile:   35   .39300       31.56432
  Effic Profile:   36   .41750       31.65406
  Effic Profile:   37   .44200       31.74975
  Effic Profile:   38   .46650       31.85133
  Effic Profile:   39   .49100       31.95866
  Effic Profile:   40   .51550       32.07156
  Effic Profile:   41   .54000       32.18975

 Profile likelihood for Density:
                 index   Density  LogLikelihood
Density Profile:    1      .00016200    29.22905893
Density Profile:    2      .00016200    29.22905893
Density Profile:    3      .00016200    29.22905893
Density Profile:    4      .00016200    29.22905893
Density Profile:    5      .00016200    29.22905893
Density Profile:    6      .00016200    29.22905893
Density Profile:    7      .00016200    29.22905893
Density Profile:    8      .00016200    29.22905893
Density Profile:    9      .00016200    29.22905893
Density Profile:   10      .00016200    29.22905893
Density Profile:   11      .00016200    29.22905893
```

Density Profile:	12	.00016200	29.22905893
Density Profile:	13	.00023050	26.66922005
Density Profile:	14	.00030311	25.21226919
Density Profile:	15	.00037572	24.45850971
Density Profile:	16	.00044833	24.02839287
Density Profile:	17	.00052094	23.75902407
Density Profile:	18	.00059355	25.84850426
Density Profile:	19	.00066616	24.52353492
Density Profile:	20	.00073877	23.60057050
Density Profile:	21	.00081138	23.28557579
Density Profile:	22	.00088399	23.54255147
Density Profile:	23	.00095660	24.12091652
Density Profile:	24	.00102921	24.79826094
Density Profile:	25	.00110182	25.46268567
Density Profile:	26	.00117443	26.07450602
Density Profile:	27	.00124704	26.62530074
Density Profile:	28	.00131965	27.11834033
Density Profile:	29	.00139226	27.55999882
Density Profile:	30	.00146487	27.95714181
Density Profile:	31	.00153748	28.31625721
Density Profile:	32	.00161009	28.64241924
Density Profile:	33	.00168270	28.94051584
Density Profile:	34	.00175531	29.21403369
Density Profile:	35	.00182792	29.46624412
Density Profile:	36	.00190053	29.69971373
Density Profile:	37	.00197314	29.91681427
Density Profile:	38	.00204575	30.11922781
Density Profile:	39	.00211836	30.30866756
Density Profile:	40	.00219097	30.48642788
Density Profile:	41	.00226358	30.65378127

F/V Endurance Depletion Exp#7- Monkfish Cooperative Survey 2009 COOKIE sweep

```
 Initial Values of parameters
    .00009                        =Mean  density of monkfish per sq ft
    .60000                        =Efficiency of trawl
 800.00000                        =K parameter for negative binomial dist
    .79900                        =Gamma parameter;  initial guess=trawl width/cell width
 Bounds on parameters
  Param #            Lower Bound                Upper Bound
           1   1.000000000000000E-007   2.000000000000000E-002
           2   5.000000000000000E-002   9.500000000000000E-001
           3   5.000000000000000E-001     2000.100000000000000
           4   8.000000000000000E-001   8.001000000000000E-001

 max # tows=          4
 Max number of hits =          4

 Value of likelihood function at initial guess:
     .000090     .600000  800.000000     .799000       39.6190825282

  Starting Pt    Density  Efficiency      K Parameter   Gamma Par  Likelihood Fc
 n
       Init Cond     .000164     .068396     .500000     .800001       14.520380
 Restart from IC     .000164     .068396     .500000     .800001       14.520380
 At 0.75 Current     .000217     .050597     .500000     .800033       14.520614
 At 1.25 Current     .000096     .122081  606.758726     .800000        9.394790
      At Current     .000097     .120066  598.376117     .800004        9.394684
 At 1.25 IniCond     .000168     .066605     .500000     .800064       14.520382
      At Current     .000164     .068228     .500000     .800063       14.520380
 At 0.75 IniCond     .000100     .116296 2000.020086     .800022        9.386830
      At Current     .000100     .116296 2000.020086     .800022        9.386830

BEST soln=      .00010      .11630  2000.02009      .80002    9.3868300530
   Ave Density/ft^2        .0000997
   Efficiency              .1162964
   K Parameter           2000.02009
   Gamma Parameter           .80002

Profile range for m=         .0000705456         .0001360520
Profile range for e=         .0792421718         .1670465201
Profile range for k=  -398821.7806384688   -59781.3312306910
Profile range for g=         .1090324463       12.9770168859

 Profile likelihood for Gamma:
                index   gamma   LogLikelihood
  Gamma Profile:    1   .50000       9.38683
  Gamma Profile:    2   .51250       9.38686
  Gamma Profile:    3   .52500       9.38687
  Gamma Profile:    4   .53750       9.38683
  Gamma Profile:    5   .55000       9.38683
  Gamma Profile:    6   .56250       9.38682
  Gamma Profile:    7   .57500       9.38683
  Gamma Profile:    8   .58750       9.38685
  Gamma Profile:    9   .60000       9.38683
  Gamma Profile:   10   .61250       9.38787
  Gamma Profile:   11   .62500       9.38770
  Gamma Profile:   12   .63750       9.38783
  Gamma Profile:   13   .65000       9.38762
  Gamma Profile:   14   .66250       9.38757
  Gamma Profile:   15   .67500       9.38759
  Gamma Profile:   16   .68750       9.38752
  Gamma Profile:   17   .70000       9.38737
  Gamma Profile:   18   .71250       9.38722
  Gamma Profile:   19   .72500       9.38711
  Gamma Profile:   20   .73750       9.38719
  Gamma Profile:   21   .75000       9.38705
  Gamma Profile:   22   .76250       9.38695
  Gamma Profile:   23   .77500       9.38685

  Gamma Profile:   24   .78750       9.38683
  Gamma Profile:   25   .80000       9.38682
  Gamma Profile:   26   .81250       9.38685
```

```
  Gamma Profile:   27   .82500        9.38687
  Gamma Profile:   28   .83750        9.38688
  Gamma Profile:   29   .85000        9.38692
  Gamma Profile:   30   .86250        9.38688
  Gamma Profile:   31   .87500        9.38696
  Gamma Profile:   32   .88750        9.38690
  Gamma Profile:   33   .90000        9.38683
  Gamma Profile:   34   .91250        9.38683
  Gamma Profile:   35   .92500        9.38683
  Gamma Profile:   36   .93750        9.38683
  Gamma Profile:   37   .95000        9.38682
  Gamma Profile:   38   .96250        9.38682
  Gamma Profile:   39   .97500        9.38690
  Gamma Profile:   40   .98750        9.38689
  Gamma Profile:   41  1.00000        9.38683
 Profile likelihood for Efficiency:
                 index   Effic   LogLikelihood
  Effic Profile:    1   .01000        9.51380
  Effic Profile:    2   .01000        9.51380
  Effic Profile:    3   .01000        9.51380
  Effic Profile:    4   .01000        9.51380
  Effic Profile:    5   .01000        9.51380
  Effic Profile:    6   .01000        9.51380
  Effic Profile:    7   .01000        9.51380
  Effic Profile:    8   .01000        9.51380
  Effic Profile:    9   .01000        9.51380
  Effic Profile:   10   .01000        9.51380
  Effic Profile:   11   .01000        9.51380
  Effic Profile:   12   .01000        9.51380
  Effic Profile:   13   .01000        9.51380
  Effic Profile:   14   .01000        9.51380
  Effic Profile:   15   .01000        9.51380
  Effic Profile:   16   .01000        9.51380
  Effic Profile:   17   .01830        9.49515
  Effic Profile:   18   .04280        9.44860
  Effic Profile:   19   .06730        9.41460
  Effic Profile:   20   .09180        9.39389
  Effic Profile:   21   .11630        9.38683
  Effic Profile:   22   .14080        9.39429
  Effic Profile:   23   .16530        9.41644
  Effic Profile:   24   .18980        9.45407
  Effic Profile:   25   .21430        9.50768
  Effic Profile:   26   .23880        9.57794
  Effic Profile:   27   .26330        9.66541
  Effic Profile:   28   .28780        9.77065
  Effic Profile:   29   .31230       14.57124
  Effic Profile:   30   .33680       14.58250
  Effic Profile:   31   .36130       14.59494
  Effic Profile:   32   .38580       14.60856
  Effic Profile:   33   .41030       14.62335
  Effic Profile:   34   .43480       14.63928
  Effic Profile:   35   .45930       14.65631
  Effic Profile:   36   .48380       14.67439
  Effic Profile:   37   .50830       14.69345
  Effic Profile:   38   .53280       11.91269
  Effic Profile:   39   .55730       11.85145
  Effic Profile:   40   .58180       12.59604
  Effic Profile:   41   .60630       12.97016

 Profile likelihood for Density:
                 index   Density  LogLikelihood
Density Profile:    1      .00001764    14.93172055
Density Profile:    2      .00001764    14.93172055
Density Profile:    3      .00001764    14.93172055
Density Profile:    4      .00001764    14.93172055
Density Profile:    5      .00001764    14.93172055
Density Profile:    6      .00001764    14.93172055
Density Profile:    7      .00001764    14.93172055
Density Profile:    8      .00001764    14.93172055
Density Profile:    9      .00001764    14.93172055
Density Profile:   10      .00001764    14.93172055
```

Density Profile:	11	.00001764	14.93172055
Density Profile:	12	.00001764	14.93172055
Density Profile:	13	.00002160	13.77880118
Density Profile:	14	.00003137	11.30256698
Density Profile:	15	.00004113	10.07538587
Density Profile:	16	.00005089	9.65490459
Density Profile:	17	.00006066	9.49482062
Density Profile:	18	.00007042	9.42840370
Density Profile:	19	.00008018	9.40027994
Density Profile:	20	.00008995	9.38939803
Density Profile:	21	.00009971	9.38683005
Density Profile:	22	.00010947	9.38840543
Density Profile:	23	.00011923	9.39388620
Density Profile:	24	.00012900	9.39665734
Density Profile:	25	.00013876	9.40190813
Density Profile:	26	.00014852	9.40701493
Density Profile:	27	.00015829	9.41181456
Density Profile:	28	.00016805	9.41683211
Density Profile:	29	.00017781	9.42129484
Density Profile:	30	.00018758	9.42564952
Density Profile:	31	.00019734	9.42980524
Density Profile:	32	.00020710	9.43368563
Density Profile:	33	.00021686	9.43732971
Density Profile:	34	.00022663	9.47706666
Density Profile:	35	.00023639	9.58334337
Density Profile:	36	.00024615	9.75074392
Density Profile:	37	.00025592	9.97286639
Density Profile:	38	.00026568	10.21479092
Density Profile:	39	.00027544	10.45613707
Density Profile:	40	.00028521	10.69062374
Density Profile:	41	.00029497	10.91506885

F/V Endurance Depletion Exp#8- Monkfish Cooperative Survey 2009 ROLLER sweep

```
 Initial Values of parameters
    .00009                 =Mean  density of monkfish per sq ft
    .60000                 =Efficiency of trawl
 800.00000                 =K parameter for negative binomial dist
    .79900                 =Gamma parameter;  initial guess=trawl width/cell width
 Bounds on parameters
  Param #             Lower Bound                Upper Bound
          1   1.000000000000000E-007   2.000000000000000E-002
          2   5.000000000000000E-002   9.500000000000000E-001
          3   5.000000000000000E-001     2000.100000000000000
          4   8.000000000000000E-001   8.001000000000000E-001

 max # tows=          5
 Max number of hits =          5

 Value of likelihood function at initial guess:
     .000090     .600000  800.000000     .799000       44.0176941086

  Starting Pt    Density  Efficiency      K Parameter   Gamma Par  Likelihood Fc
 n
       Init Cond     .000335     .050000     .500000     .800000       20.418132
 Restart from IC     .000335     .050000     .500000     .800000       20.418132
 At 0.75 Current     .000336     .050000     .500000     .800029       20.418133
 At 1.25 Current     .000333     .050009 1974.408224     .800000       12.834802
      At Current     .000333     .050005 1976.180254     .800000       12.834715
 At 1.25 IniCond     .000336     .050000     .500000     .800096       20.418134
      At Current     .000336     .050000     .500000     .800096       20.418134
 At 0.75 IniCond     .000333     .050000 1991.802224     .800000       12.834665
      At Current     .000333     .050000 1994.905678     .800000       12.834636

BEST soln=      .00033      .05000  1994.90568      .80000   12.8346363243
   Ave Density/ft^2        .0003328
   Efficiency              .0500001
   K Parameter           1994.90568
   Gamma Parameter           .80000

Profile range for m=         .0002611375         .0004168065
Profile range for e=         .0383579543         .0634661256
Profile range for k=********************   -53765.7363895542
Profile range for g=        -.3423246137        2.3707364499

 Profile likelihood for Gamma:
                index   gamma   LogLikelihood
  Gamma Profile:    1   .50000       12.70805
  Gamma Profile:    2   .51250       12.71268
  Gamma Profile:    3   .52500       12.71737
  Gamma Profile:    4   .53750       12.72212
  Gamma Profile:    5   .55000       12.72693
  Gamma Profile:    6   .56250       12.73177
  Gamma Profile:    7   .57500       12.73670
  Gamma Profile:    8   .58750       12.74168
  Gamma Profile:    9   .60000       12.74670
  Gamma Profile:   10   .61250       12.75178
  Gamma Profile:   11   .62500       12.75691
  Gamma Profile:   12   .63750       12.76210
  Gamma Profile:   13   .65000       12.76738
  Gamma Profile:   14   .66250       12.77266
  Gamma Profile:   15   .67500       12.77800
  Gamma Profile:   16   .68750       12.78340
  Gamma Profile:   17   .70000       12.78890
  Gamma Profile:   18   .71250       12.79442
  Gamma Profile:   19   .72500       12.80006
  Gamma Profile:   20   .73750       12.80569
  Gamma Profile:   21   .75000       12.81136
  Gamma Profile:   22   .76250       12.81709
  Gamma Profile:   23   .77500       12.82287
  Gamma Profile:   24   .78750       12.82875
  Gamma Profile:   25   .80000       12.83464
  Gamma Profile:   26   .81250       12.84063
  Gamma Profile:   27   .82500       12.84665
```

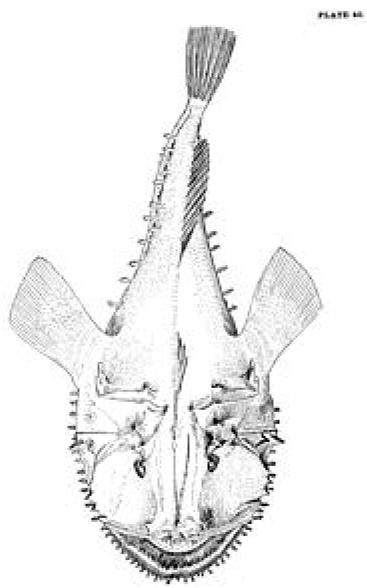
```
  Gamma Profile:   28   .83750       12.85275
  Gamma Profile:   29   .85000       12.85890
  Gamma Profile:   30   .86250       12.86508
  Gamma Profile:   31   .87500       12.87134
  Gamma Profile:   32   .88750       12.87766
  Gamma Profile:   33   .90000       12.88402
  Gamma Profile:   34   .91250       12.89044
  Gamma Profile:   35   .92500       12.89697
  Gamma Profile:   36   .93750       12.90349
  Gamma Profile:   37   .95000       12.91007
  Gamma Profile:   38   .96250       12.91672
  Gamma Profile:   39   .97500       12.92345
  Gamma Profile:   40   .98750       12.93021
  Gamma Profile:   41  1.00000       12.93707
 Profile likelihood for Efficiency:
                 index   Effic   LogLikelihood
  Effic Profile:    1   .01000       12.60228
  Effic Profile:    2   .01000       12.60228
  Effic Profile:    3   .01000       12.60228
  Effic Profile:    4   .01000       12.60228
  Effic Profile:    5   .01000       12.60228
  Effic Profile:    6   .01000       12.60228
  Effic Profile:    7   .01000       12.60228
  Effic Profile:    8   .01000       12.60228
  Effic Profile:    9   .01000       12.60228
  Effic Profile:   10   .01000       12.60228
  Effic Profile:   11   .01000       12.60228
  Effic Profile:   12   .01000       12.60228
  Effic Profile:   13   .01000       12.60228
  Effic Profile:   14   .01000       12.60228
  Effic Profile:   15   .01000       12.60228
  Effic Profile:   16   .01000       12.60228
  Effic Profile:   17   .01000       12.60228
  Effic Profile:   18   .01000       12.60228
  Effic Profile:   19   .01000       12.60308
  Effic Profile:   20   .02550       12.67566
  Effic Profile:   21   .05000       12.83463
  Effic Profile:   22   .07450       13.04949
  Effic Profile:   23   .09900       13.32233
  Effic Profile:   24   .12350       13.65665
  Effic Profile:   25   .14800       14.05521
  Effic Profile:   26   .17250       20.48278
  Effic Profile:   27   .19700       20.50371
  Effic Profile:   28   .22150       20.52764
  Effic Profile:   29   .24600       20.55473
  Effic Profile:   30   .27050       20.58515
  Effic Profile:   31   .29500       20.61907
  Effic Profile:   32   .31950       20.65668
  Effic Profile:   33   .34400       20.69817
  Effic Profile:   34   .36850       20.74375
  Effic Profile:   35   .39300       20.79365
  Effic Profile:   36   .41750       20.84809
  Effic Profile:   37   .44200       20.90732
  Effic Profile:   38   .46650       20.97162
  Effic Profile:   39   .49100       21.04127
  Effic Profile:   40   .51550       21.11659
  Effic Profile:   41   .54000       21.19792

 Profile likelihood for Density:
                 index   Density  LogLikelihood
Density Profile:    1      .00006528    17.46294046
Density Profile:    2      .00006528    17.46294046
Density Profile:    3      .00006528    17.46294046
Density Profile:    4      .00006528    17.46294046
Density Profile:    5      .00006528    17.46294046
Density Profile:    6      .00006528    17.46294046
Density Profile:    7      .00006528    17.46294046
Density Profile:    8      .00006528    17.46294046
Density Profile:    9      .00006528    17.46294046
Density Profile:   10      .00006528    17.46294046
Density Profile:   11      .00006528    17.46294046
```

Density Profile:	12	.00006617	17.36979120
Density Profile:	13	.00009579	15.14080913
Density Profile:	14	.00012542	14.08811621
Density Profile:	15	.00015505	13.55803238
Density Profile:	16	.00018468	13.27897683
Density Profile:	17	.00021431	16.95863739
Density Profile:	18	.00024394	15.49417561
Density Profile:	19	.00027356	14.10377757
Density Profile:	20	.00030319	13.13309298
Density Profile:	21	.00033282	12.83463630
Density Profile:	22	.00036245	13.09709285
Density Profile:	23	.00039208	13.81427396
Density Profile:	24	.00042171	14.62438983
Density Profile:	25	.00045133	15.34839076
Density Profile:	26	.00048096	15.97379696
Density Profile:	27	.00051059	16.51280441
Density Profile:	28	.00054022	16.98068720
Density Profile:	29	.00056985	17.39070907
Density Profile:	30	.00059948	17.75348072
Density Profile:	31	.00062910	18.07731863
Density Profile:	32	.00065873	18.36874980
Density Profile:	33	.00068836	18.63286670
Density Profile:	34	.00071799	18.87377013
Density Profile:	35	.00074762	19.09474868
Density Profile:	36	.00077725	19.29847030
Density Profile:	37	.00080687	19.48712362
Density Profile:	38	.00083650	19.66255391
Density Profile:	39	.00086613	19.82626470
Density Profile:	40	.00089576	19.97956657
Density Profile:	41	.00092539	20.12354259

VERY LARGE FILE
OVER 100 PAGES
LOOK BUT PLEASE DO NOT PRINT

Final Runs
Includes model Survey and Catch length frequency fits
5/14/2010

Southern Demersal Working Group

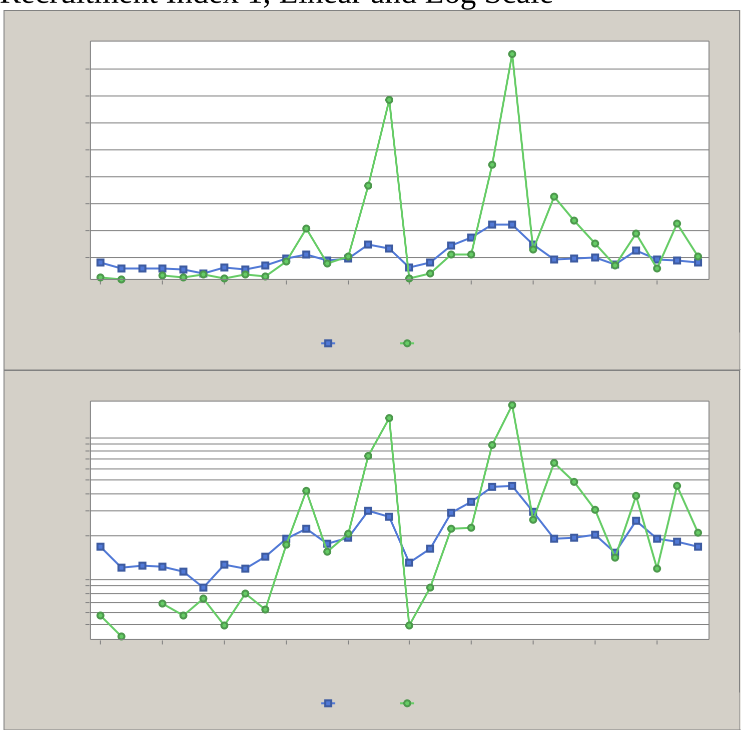


50th SAW/SARC

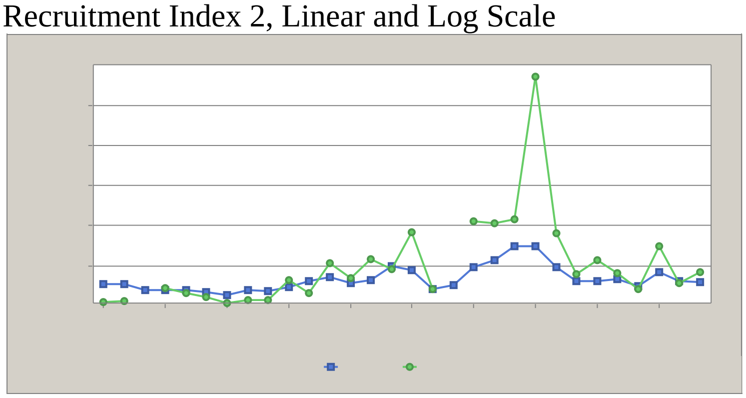
Northern Management Area Final Run 8

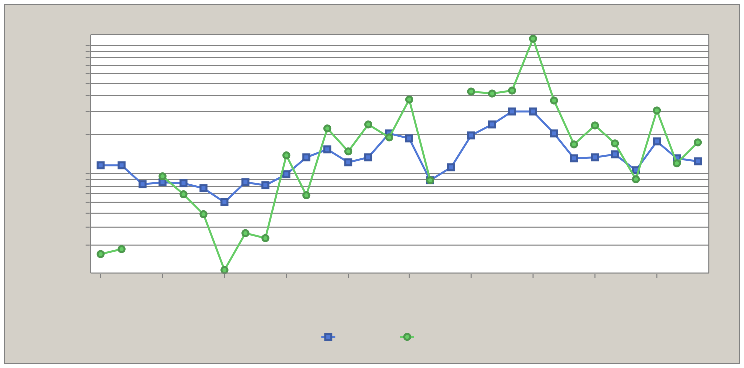
Recruitment Indices, Group Linear and Log Scale, 1 Index per Line (12 Plots)
Adult Indices, Group Linear and Log Scale, 1 Index per Line (8 Plots)
Survey Length Frequencies (210 Plots)
Catch Numbers, Catch Length Frequency (30 Plots)
Observed vs. Predicted Catch Weight (1 Plot)
Selectivity (1 Plots)
4-Plot: Population and Catch Numbers (60 Plots)
Fmult, Age 1 Recruitment, Observed vs. Predicted Catch Weight, and Total Biomass: Group 2 per Line (4 Plots)

Recruitment Index 1, Linear and Log Scale

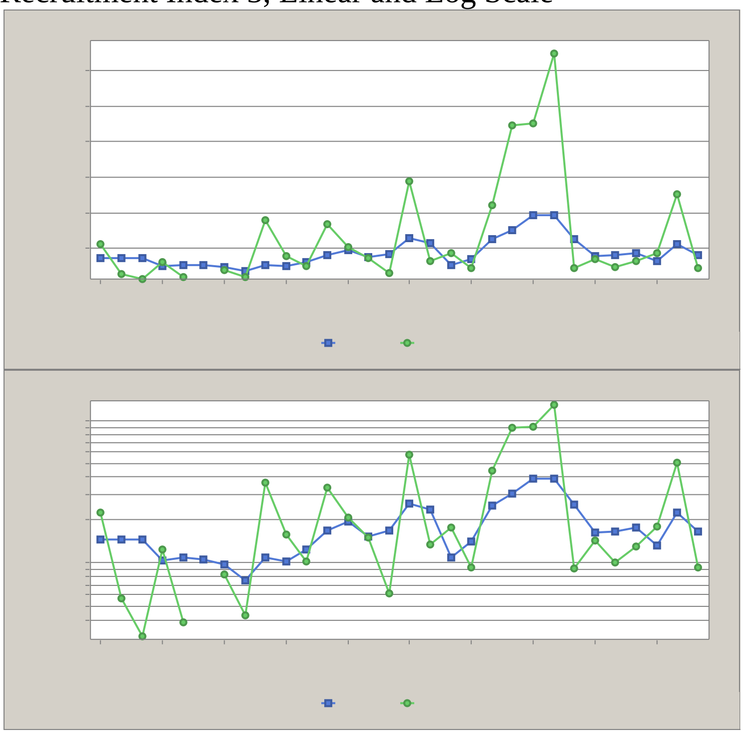


Recruitment Index 2, Linear and Log Scale

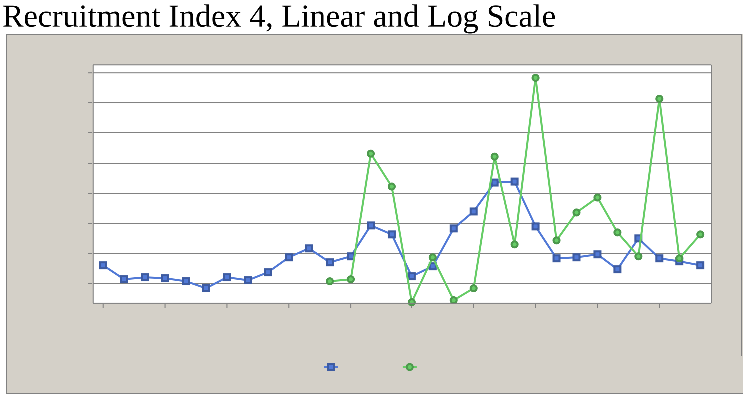


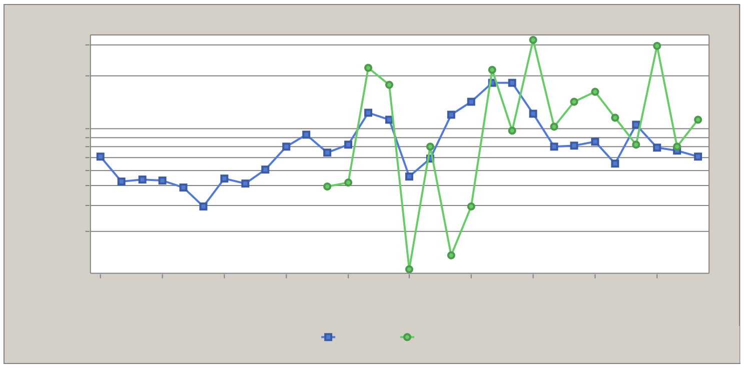


Recruitment Index 3, Linear and Log Scale

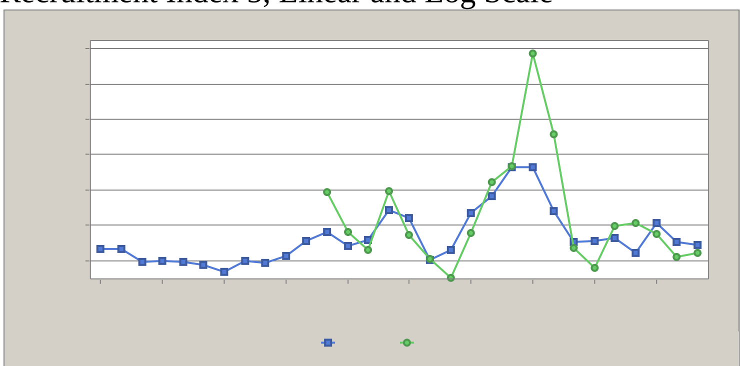


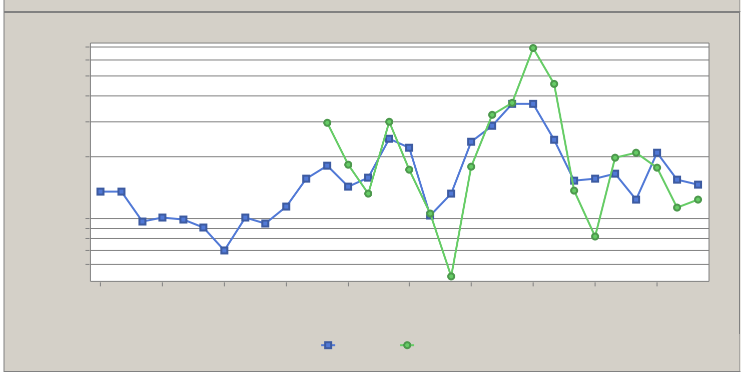
Recruitment Index 4, Linear and Log Scale



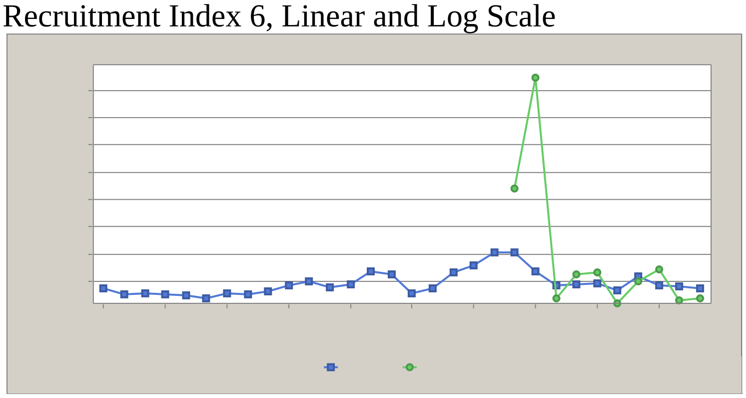


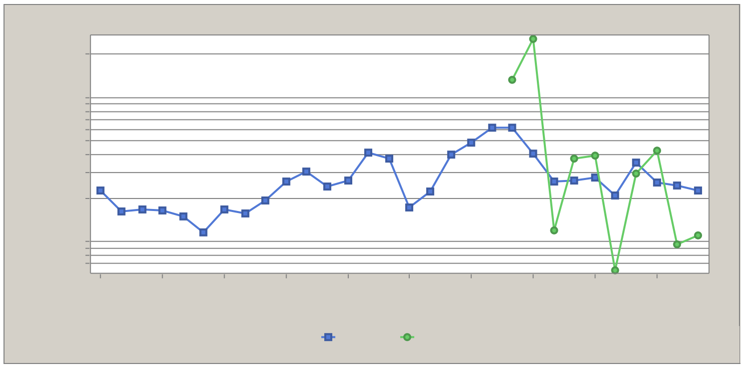
Recruitment Index 5, Linear and Log Scale



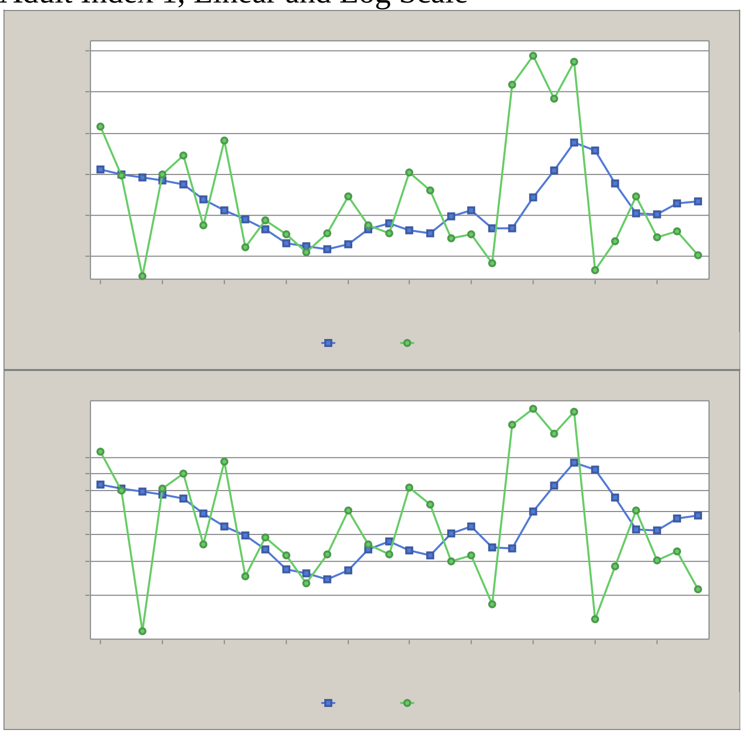


Recruitment Index 6, Linear and Log Scale

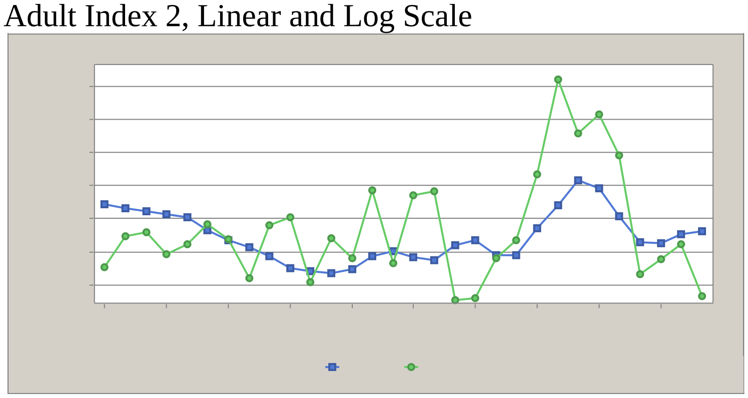


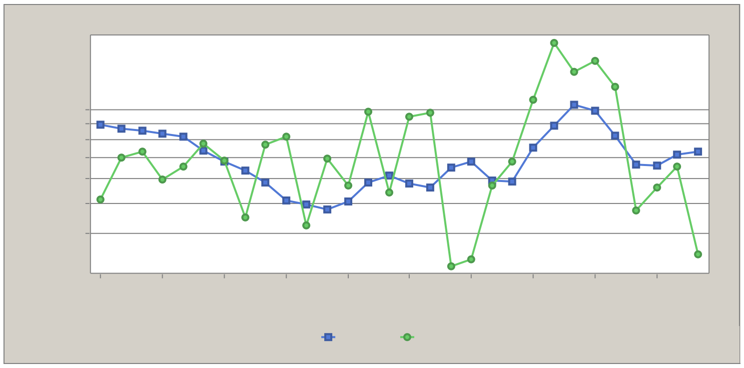


Adult Index 1, Linear and Log Scale

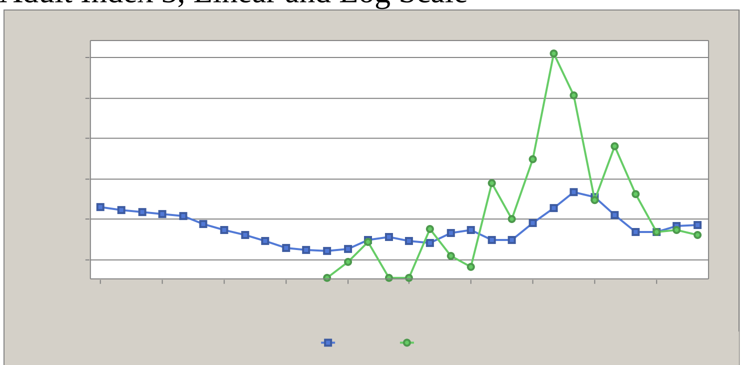


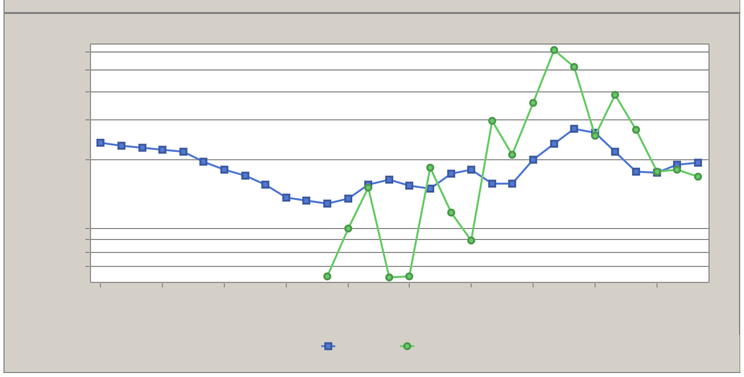
Adult Index 2, Linear and Log Scale



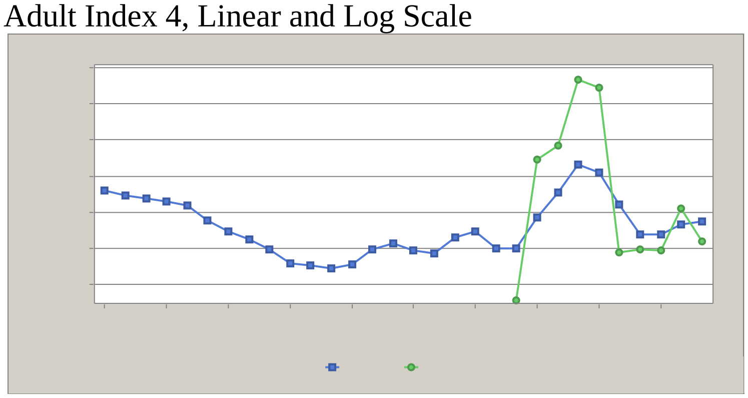


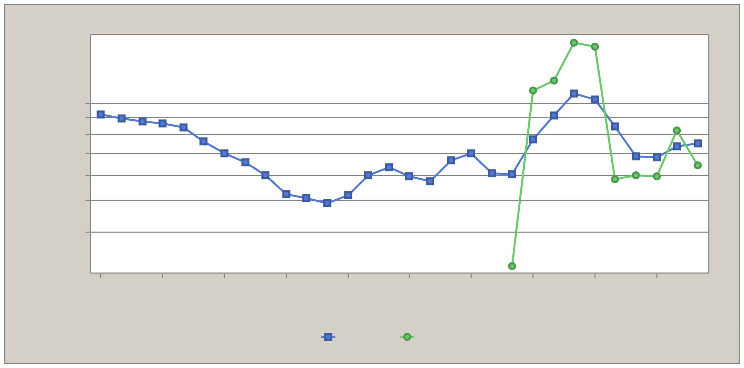
Adult Index 3, Linear and Log Scale



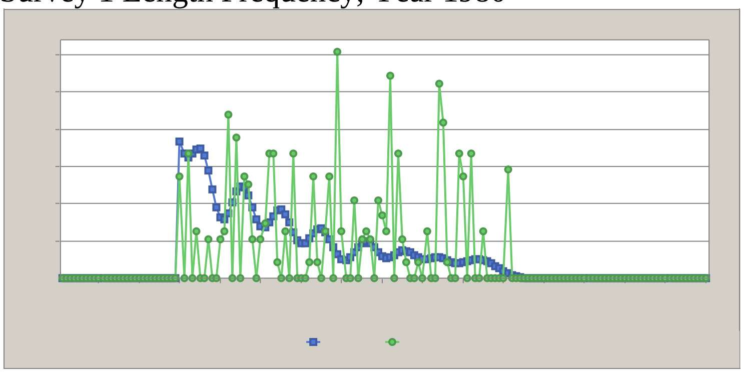


Adult Index 4, Linear and Log Scale

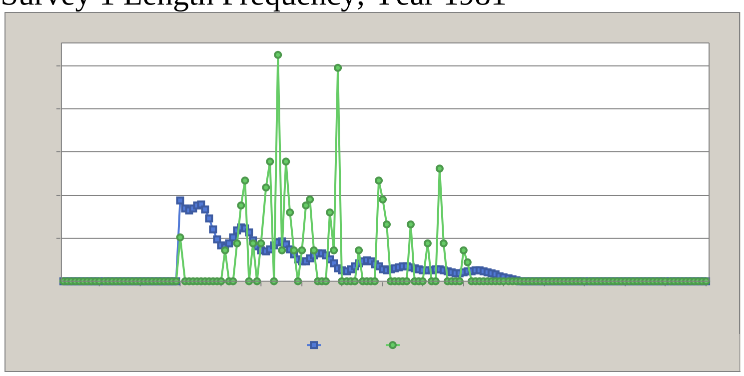




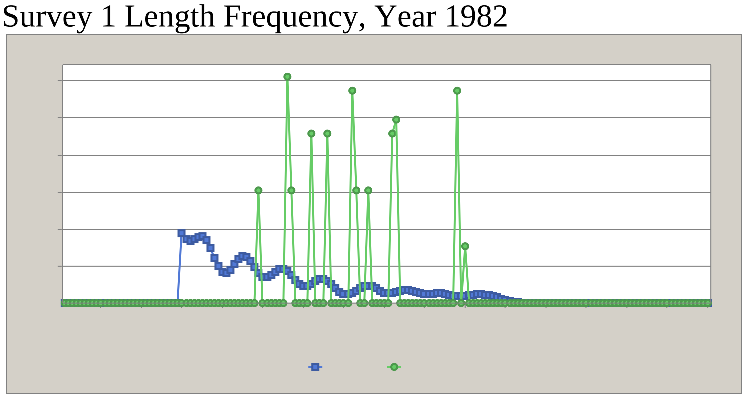
Survey 1 Length Frequency, Year 1980



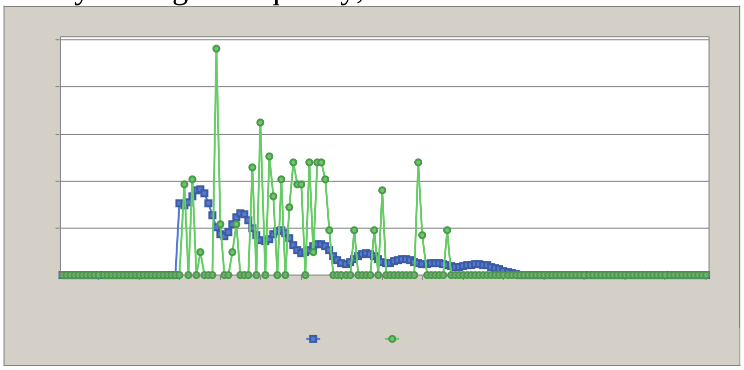
Survey 1 Length Frequency, Year 1981



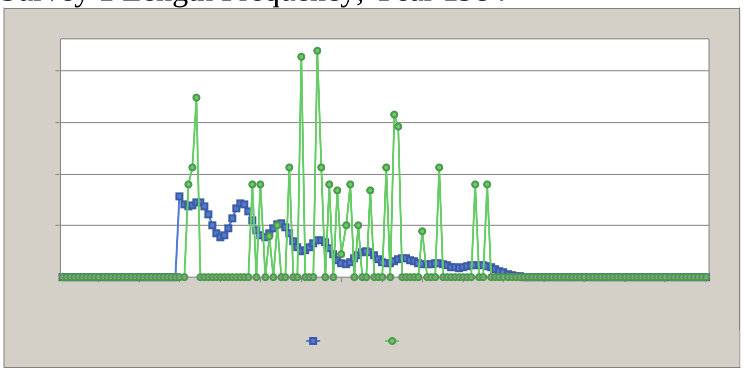
Survey 1 Length Frequency, Year 1982



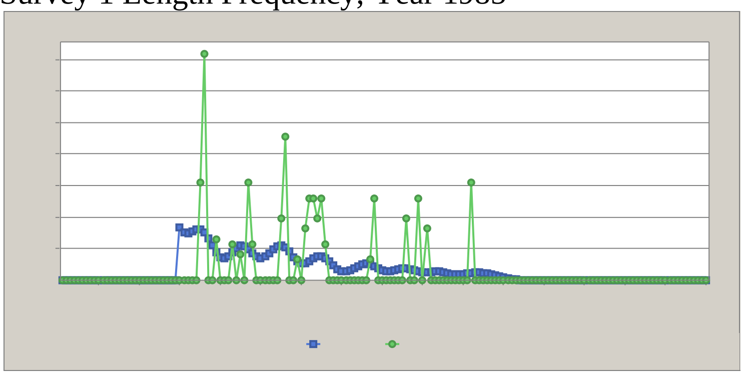
Survey 1 Length Frequency, Year 1983



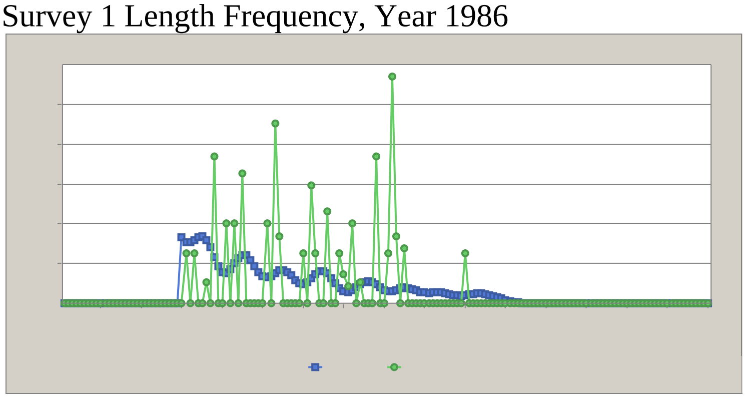
Survey 1 Length Frequency, Year 1984



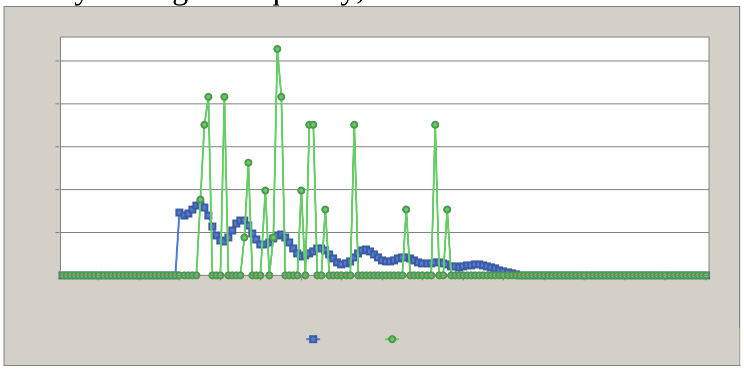
Survey 1 Length Frequency, Year 1985



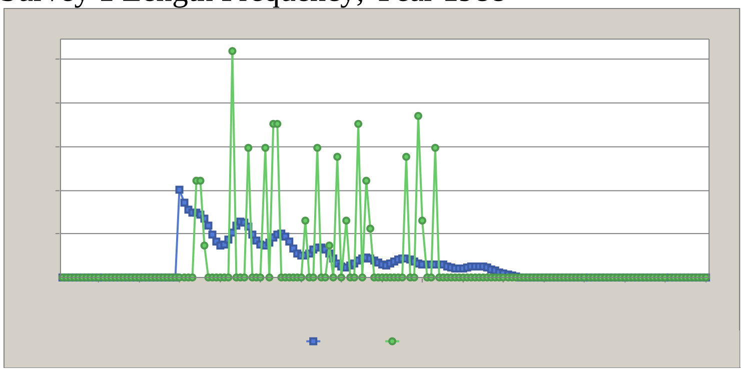
Survey 1 Length Frequency, Year 1986



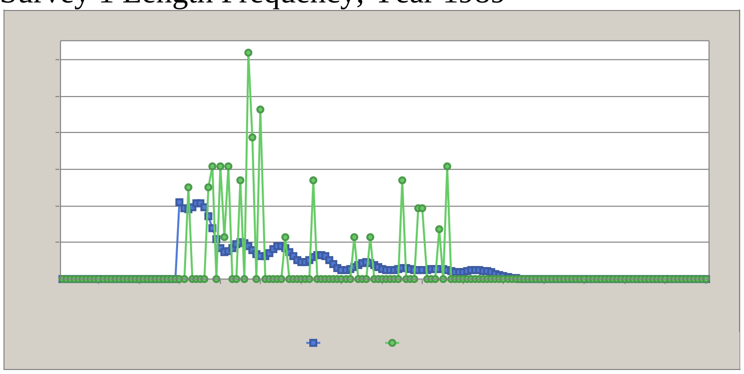
Survey 1 Length Frequency, Year 1987



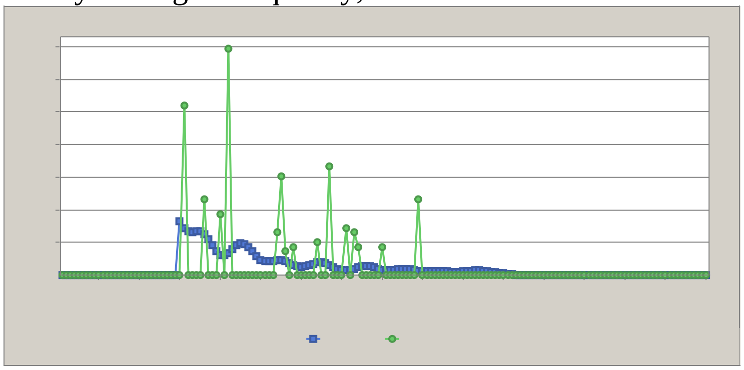
Survey 1 Length Frequency, Year 1988



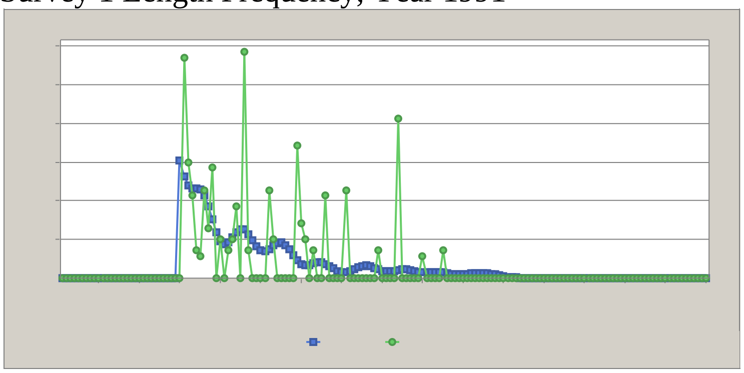
Survey 1 Length Frequency, Year 1989



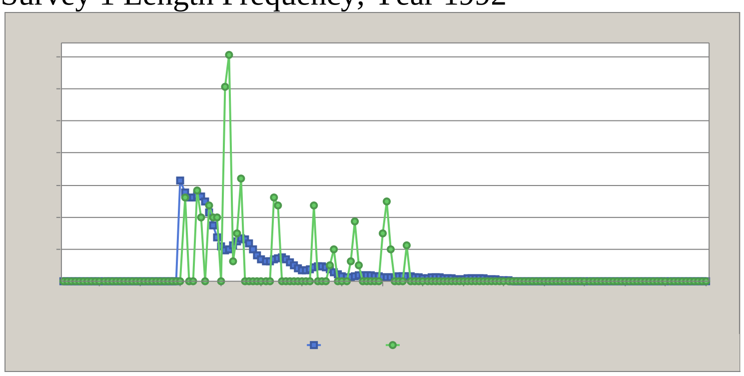
Survey 1 Length Frequency, Year 1990



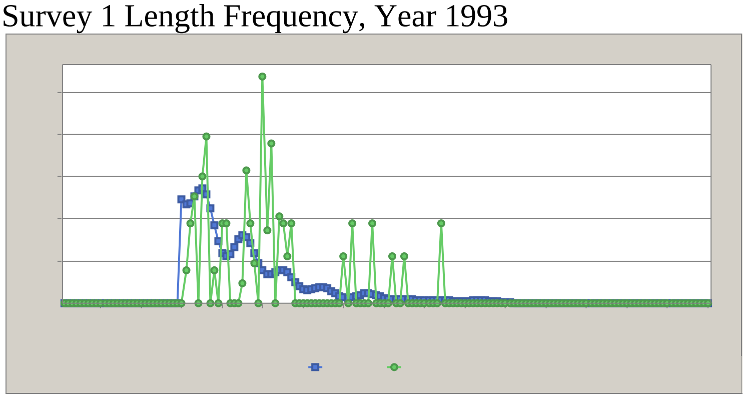
Survey 1 Length Frequency, Year 1991



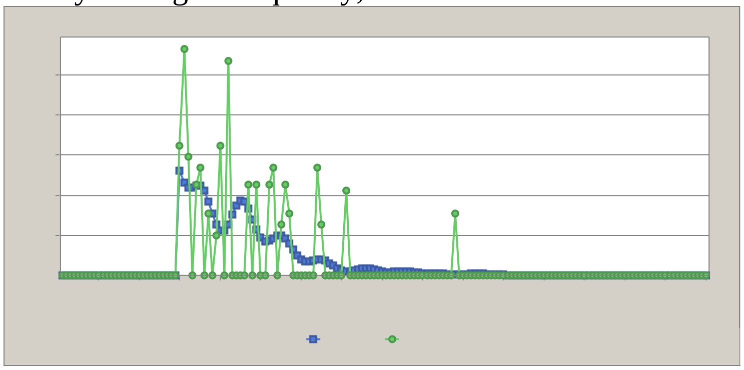
Survey 1 Length Frequency, Year 1992



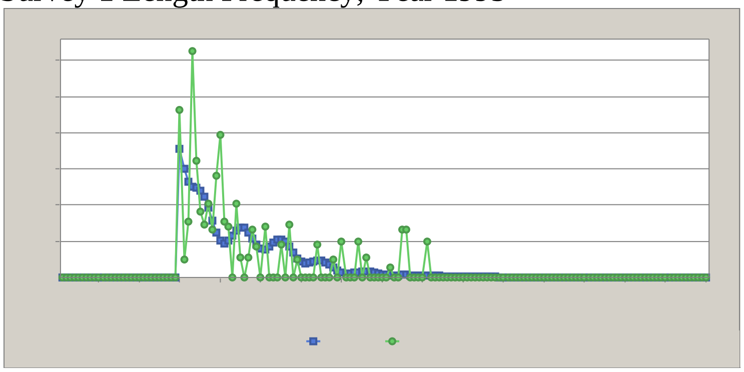
Survey 1 Length Frequency, Year 1993



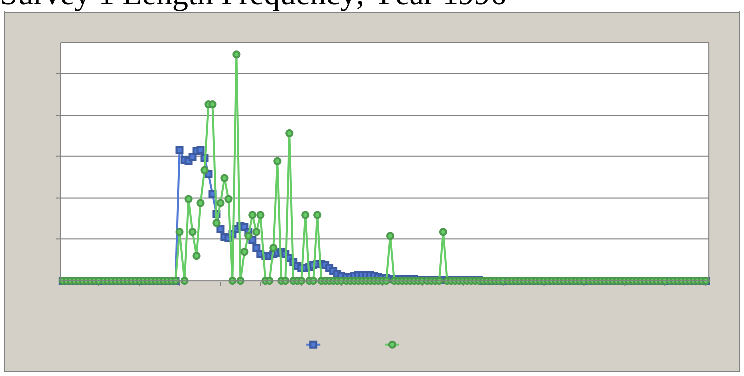
Survey 1 Length Frequency, Year 1994



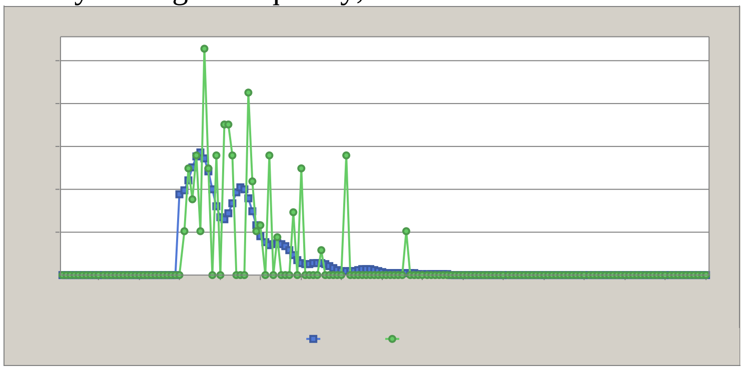
Survey 1 Length Frequency, Year 1995



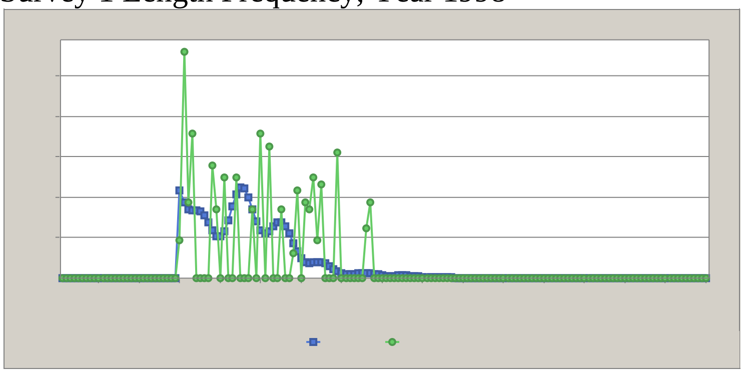
Survey 1 Length Frequency, Year 1996



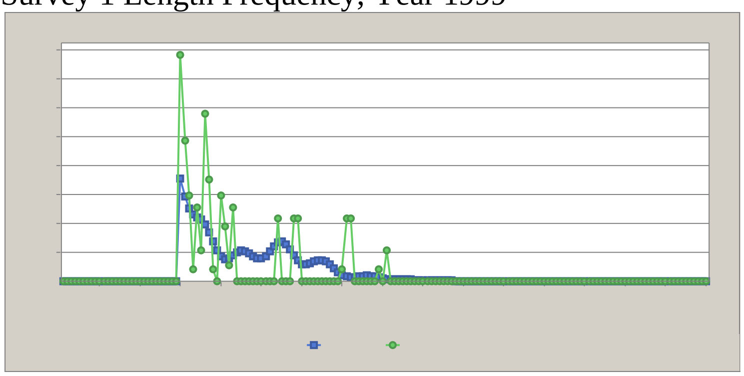
Survey 1 Length Frequency, Year 1997



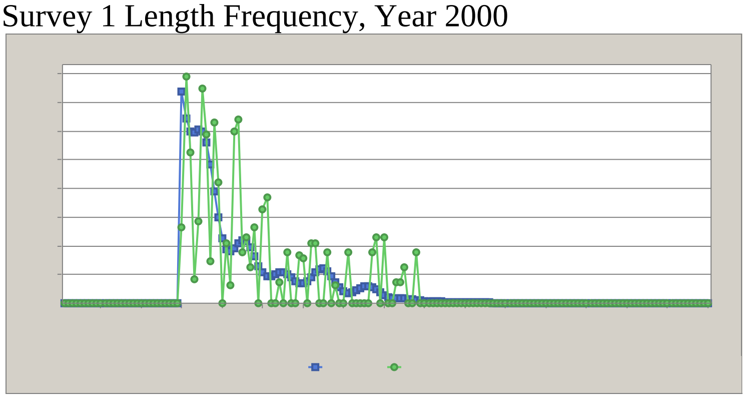
Survey 1 Length Frequency, Year 1998



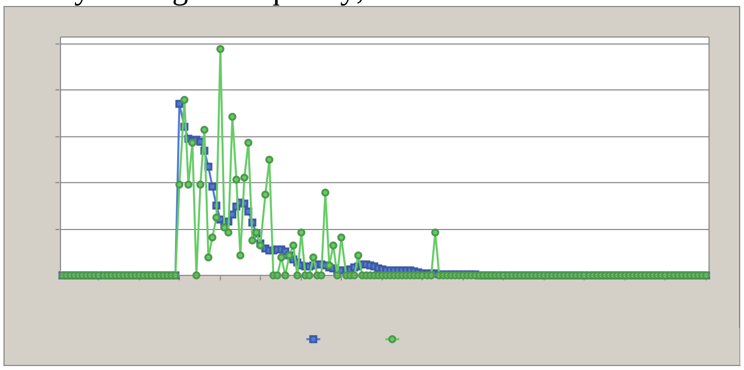
Survey 1 Length Frequency, Year 1999



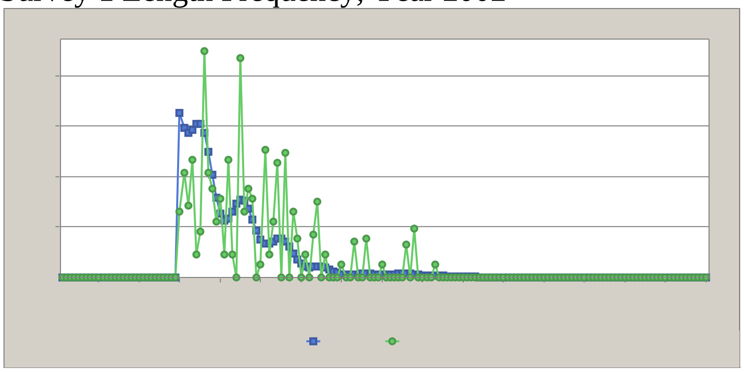
Survey 1 Length Frequency, Year 2000



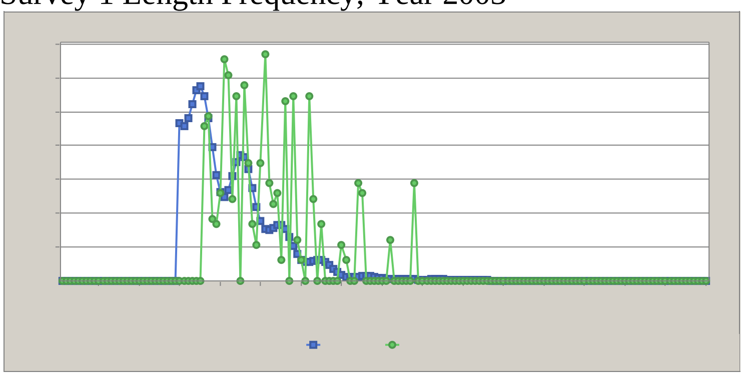
Survey 1 Length Frequency, Year 2001



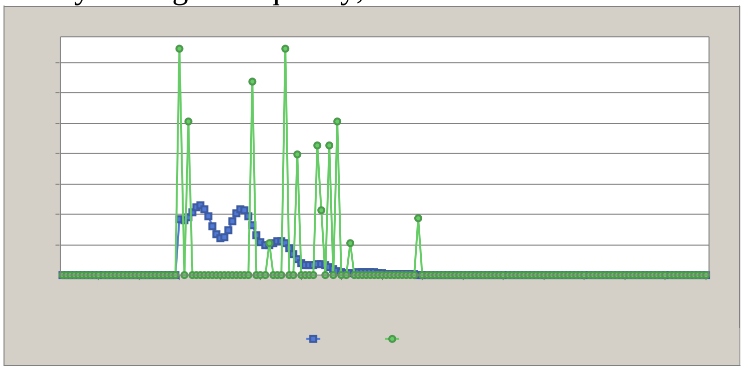
Survey 1 Length Frequency, Year 2002



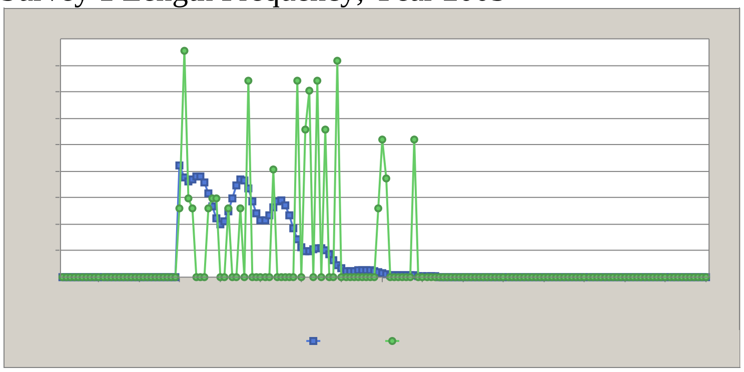
Survey 1 Length Frequency, Year 2003



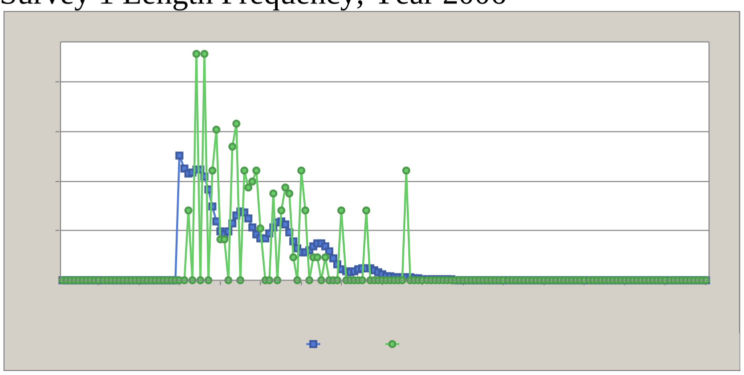
Survey 1 Length Frequency, Year 2004



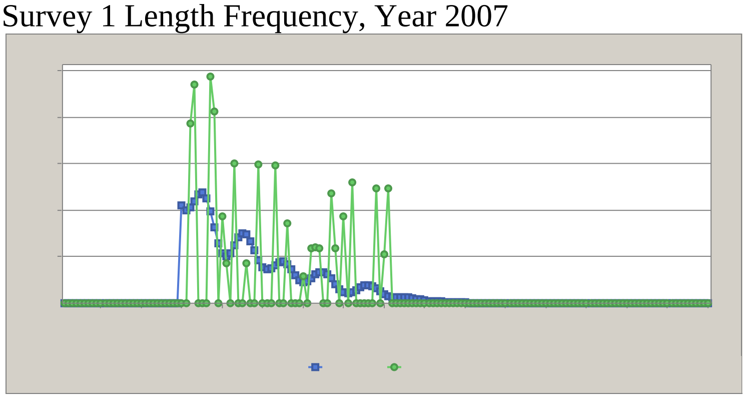
Survey 1 Length Frequency, Year 2005



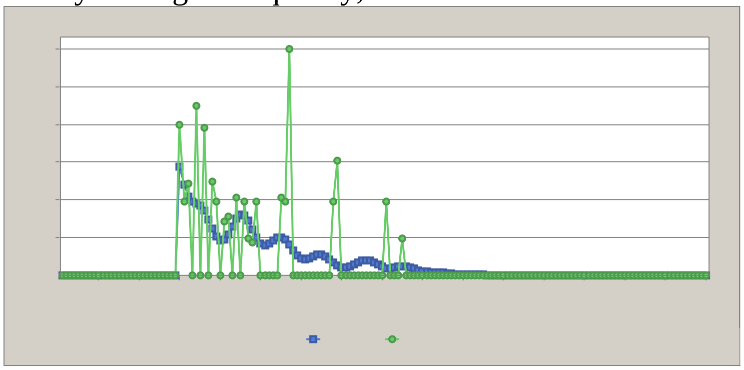
Survey 1 Length Frequency, Year 2006



Survey 1 Length Frequency, Year 2007



Survey 1 Length Frequency, Year 2008



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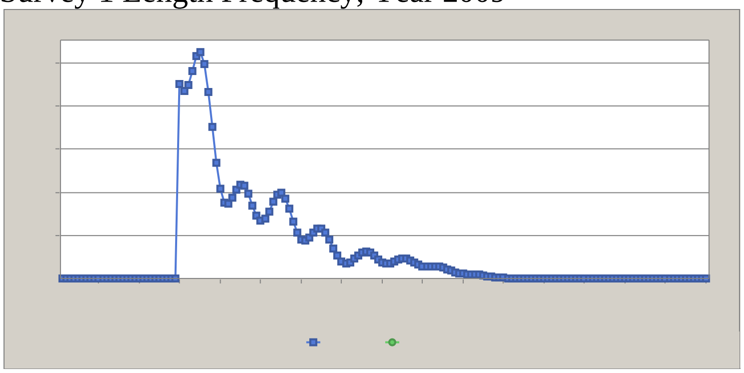
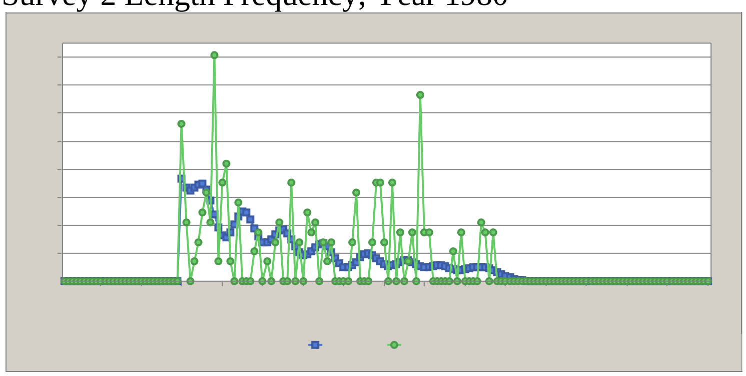
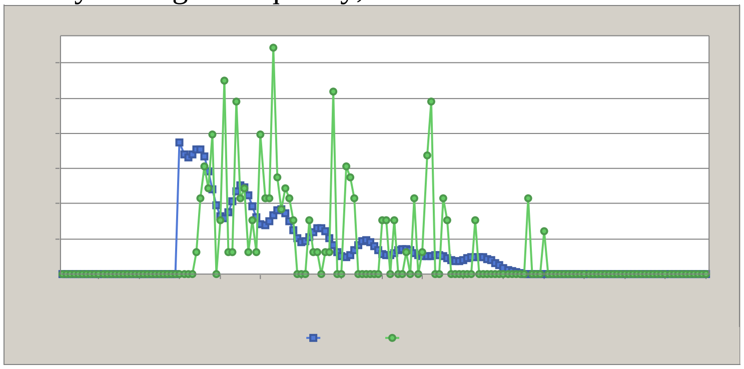


Survey 2 Length Frequency, Year 1980



Survey 2 Length Frequency, Year 1981



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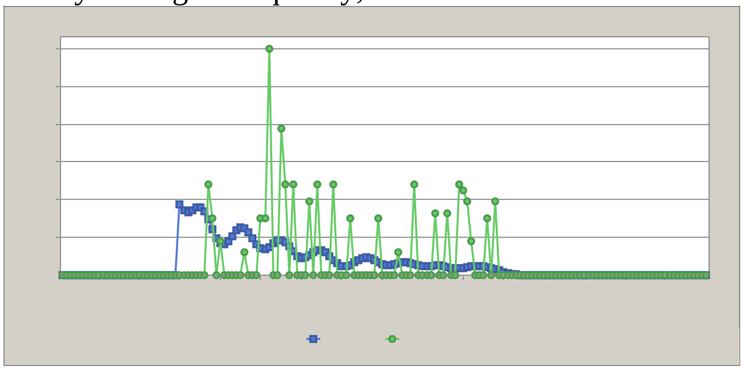

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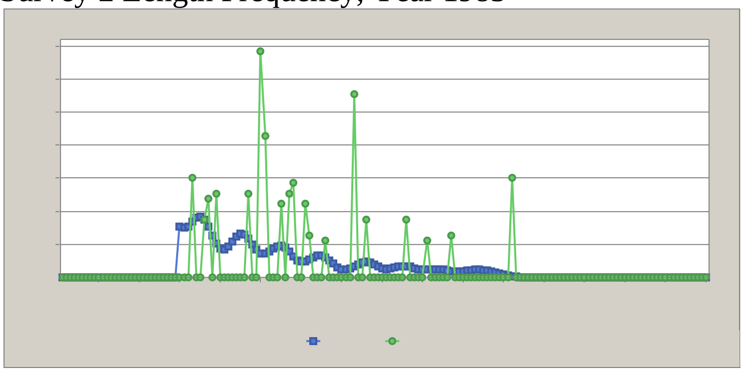

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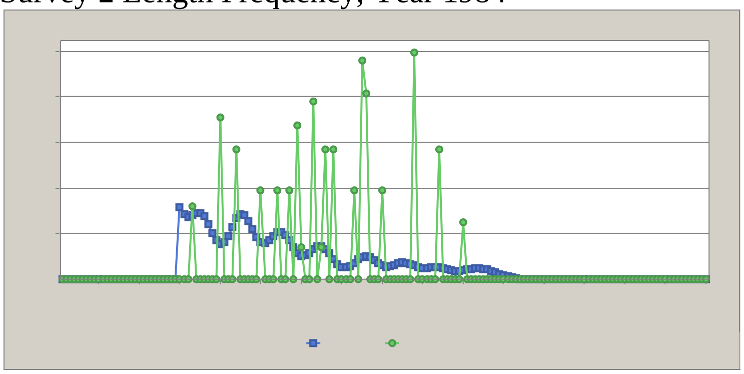

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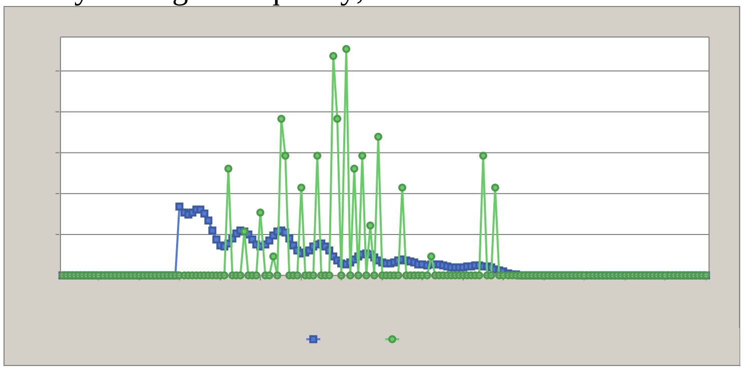

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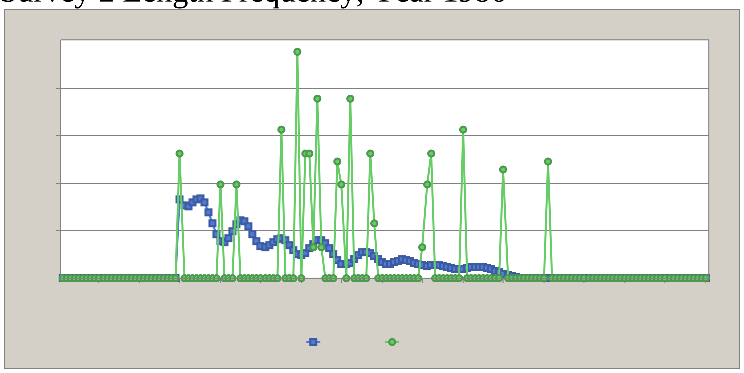
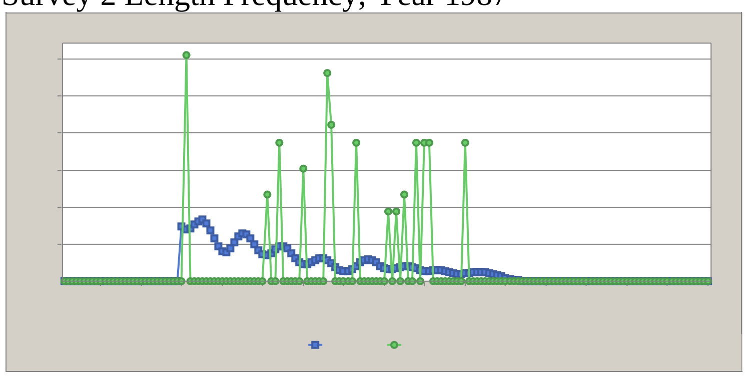


Survey 2 Length Frequency, Year 1987



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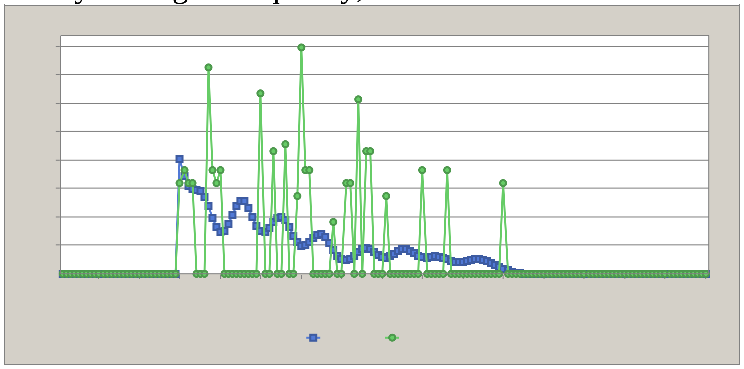

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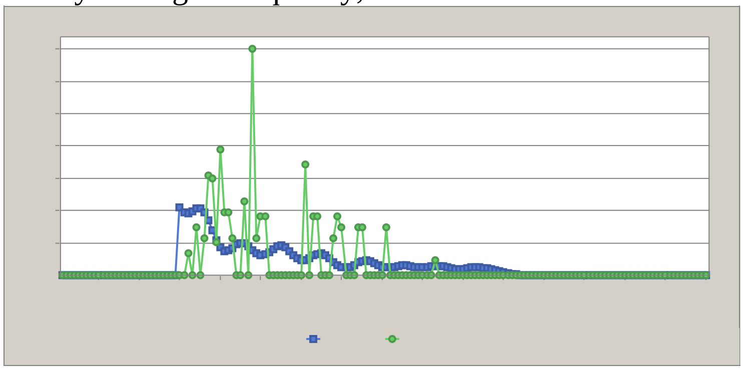

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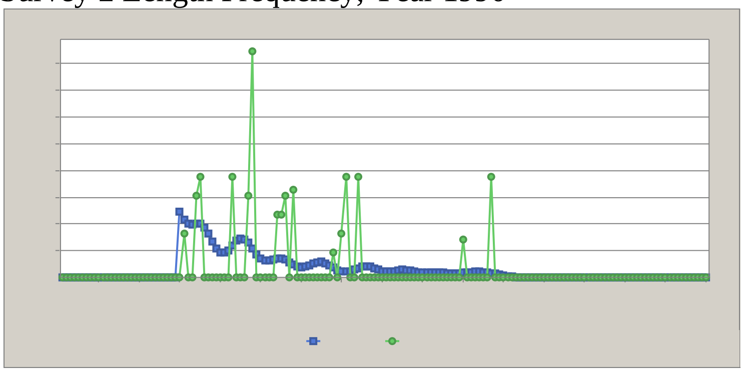
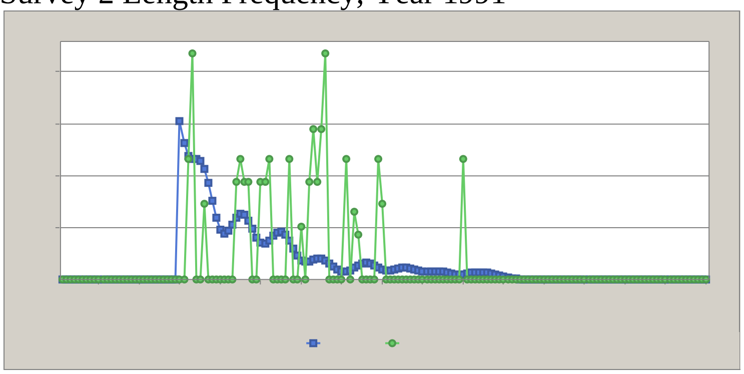


Survey 2 Length Frequency, Year 1991



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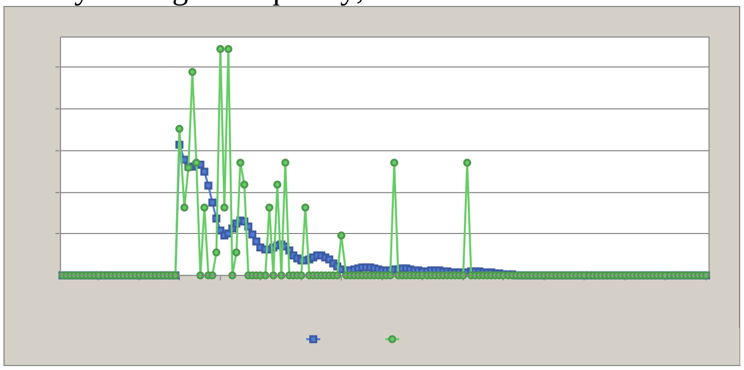

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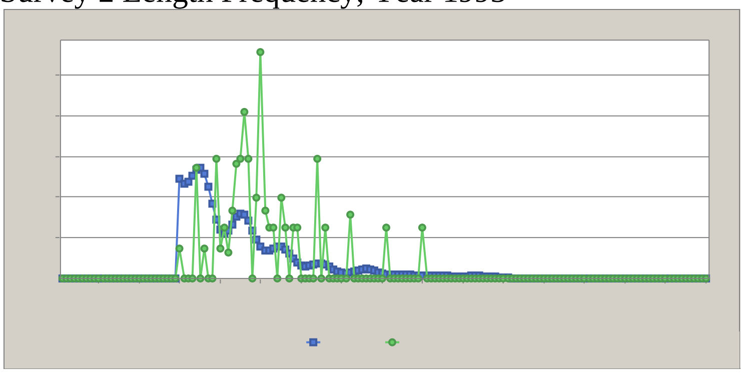
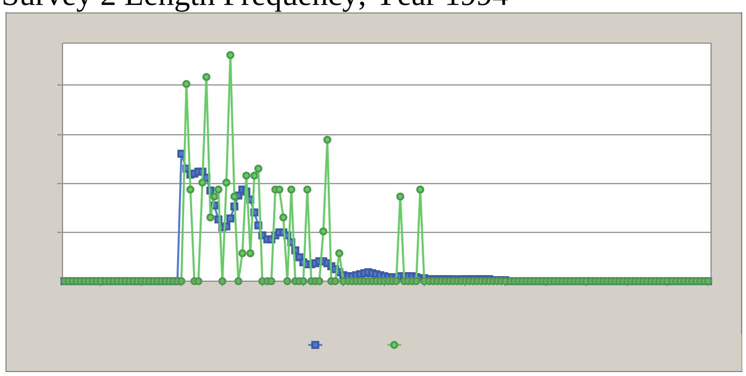
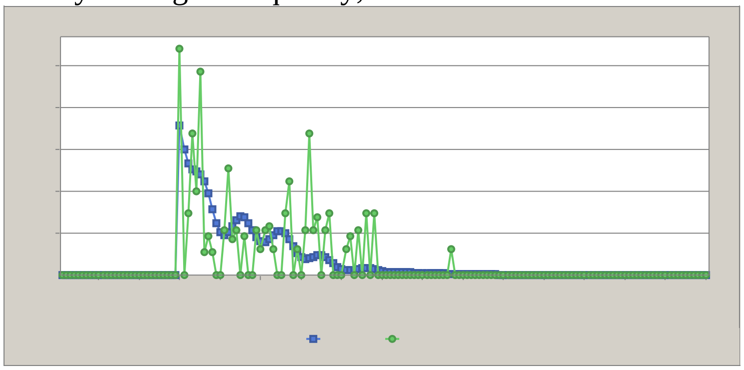


Survey 2 Length Frequency, Year 1994



Survey 2 Length Frequency, Year 1995



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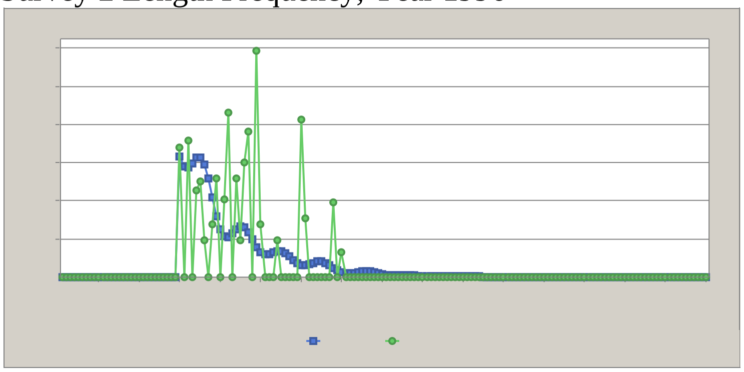
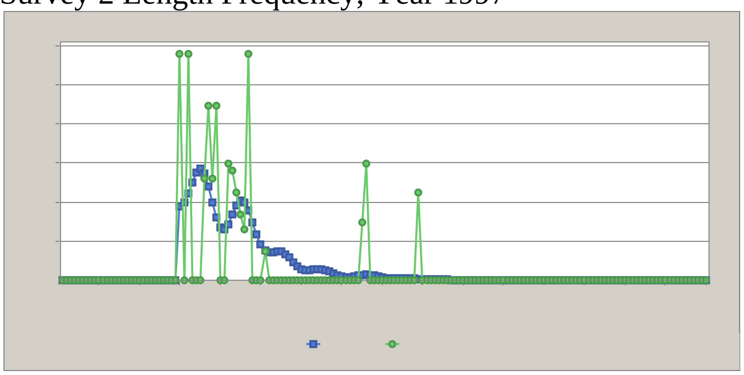
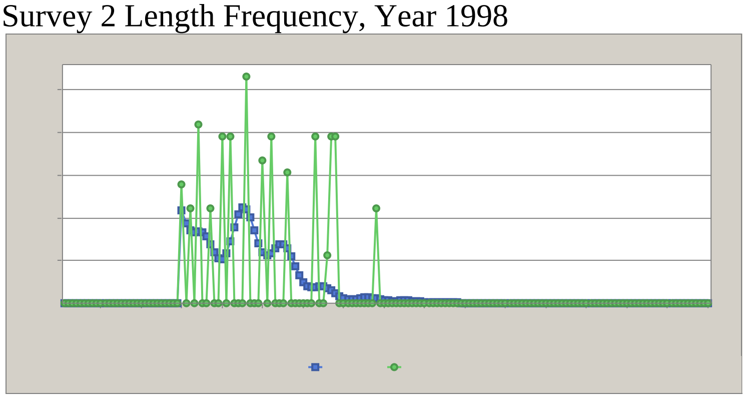


Survey 2 Length Frequency, Year 1997



Survey 2 Length Frequency, Year 1998



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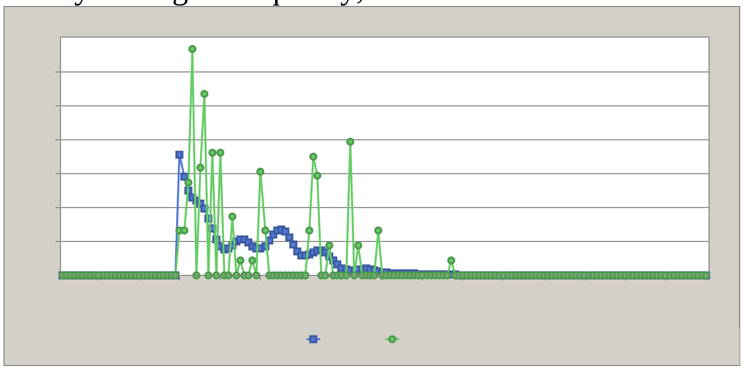

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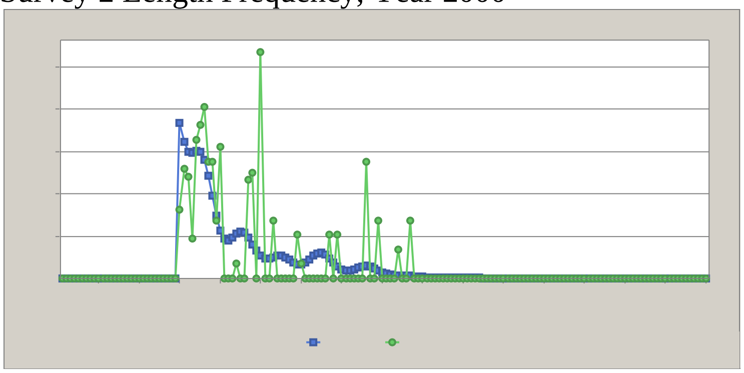
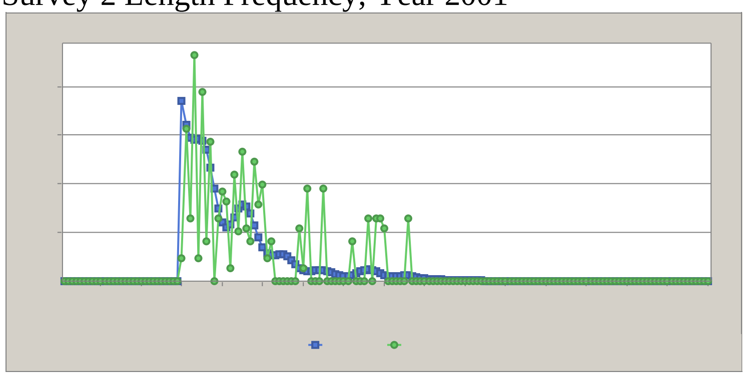


Survey 2 Length Frequency, Year 2001



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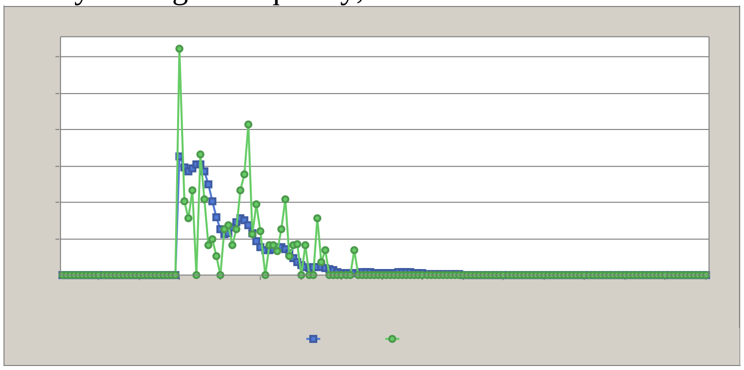

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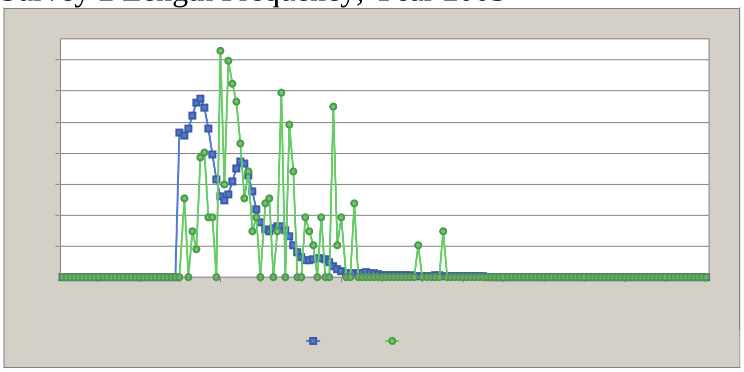

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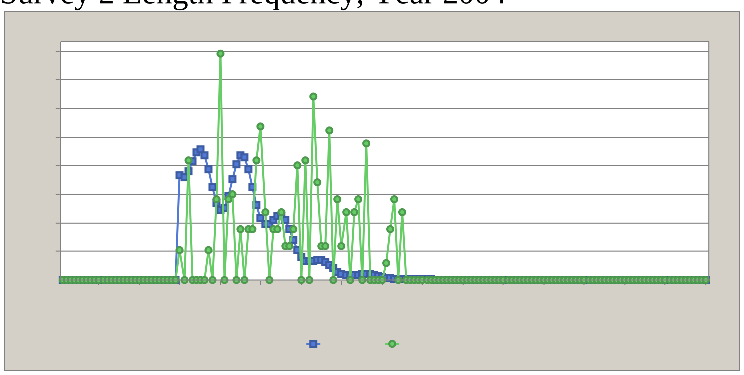

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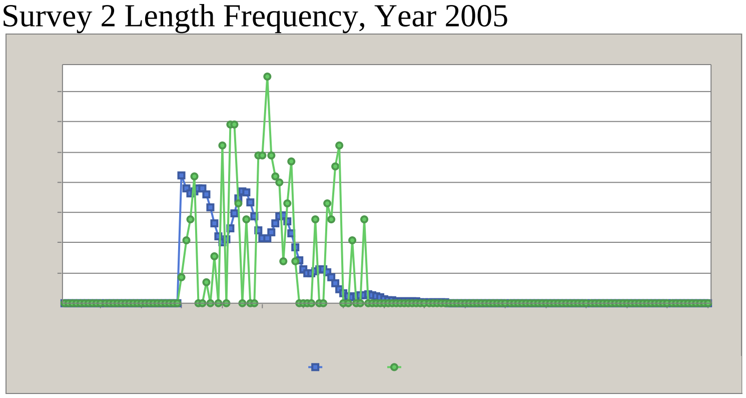

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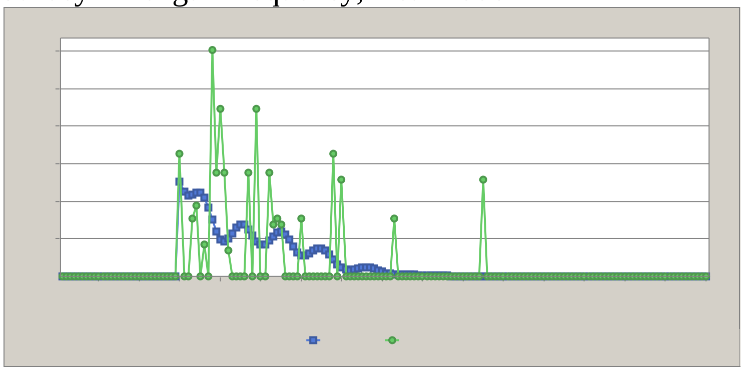

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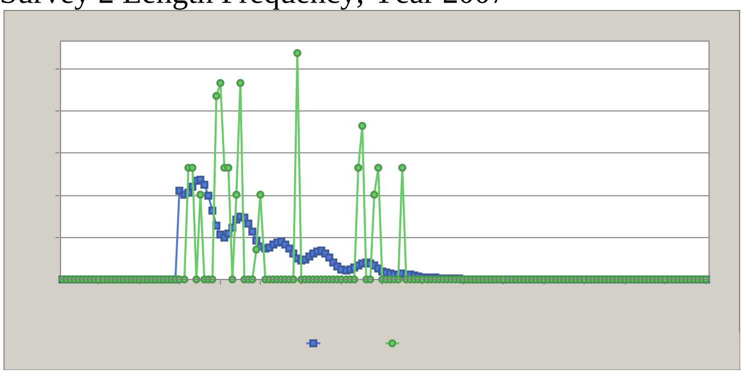
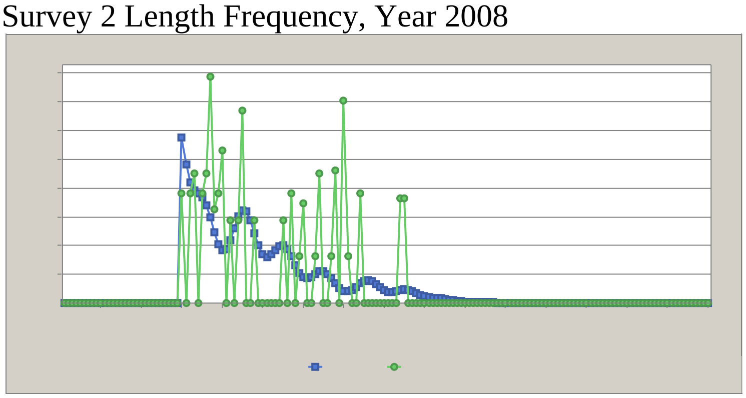


Survey 2 Length Frequency, Year 2008



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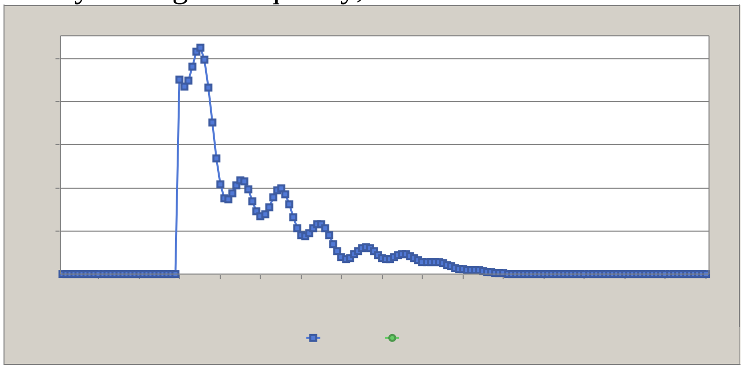
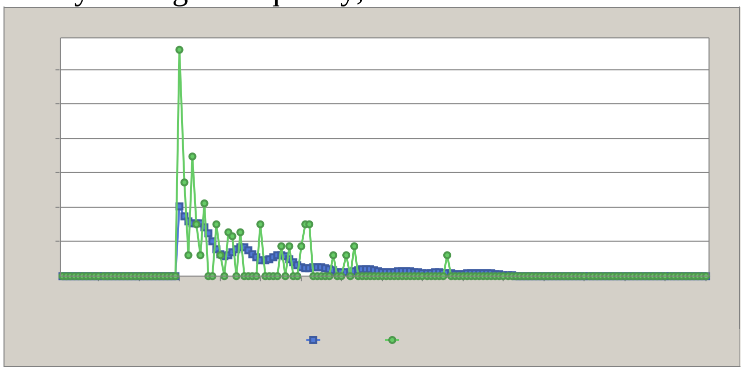


Survey 3 Length Frequency, Year 1991



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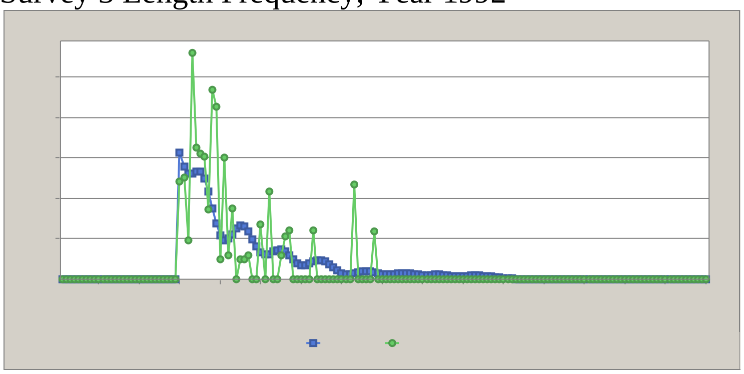

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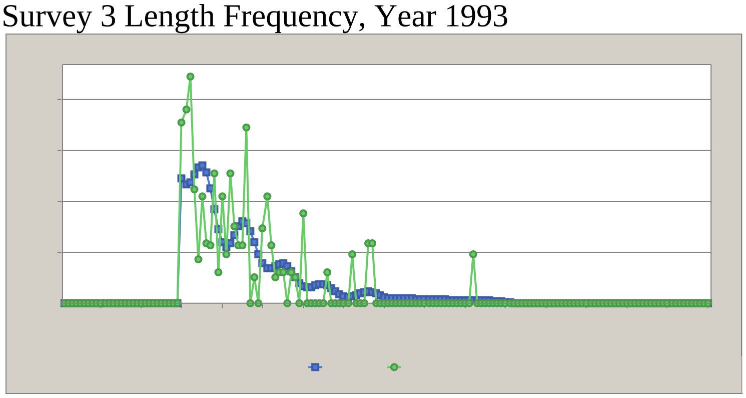

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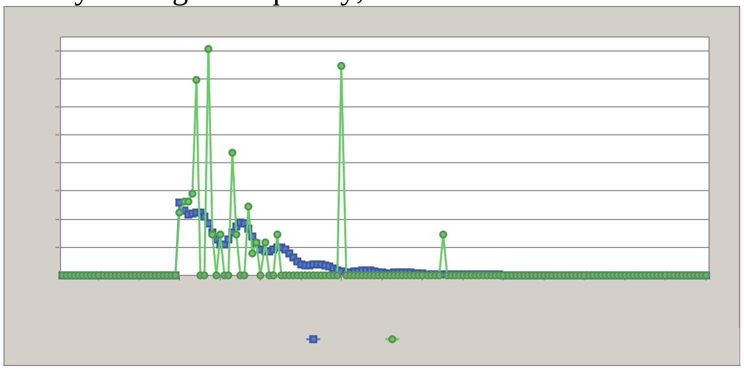

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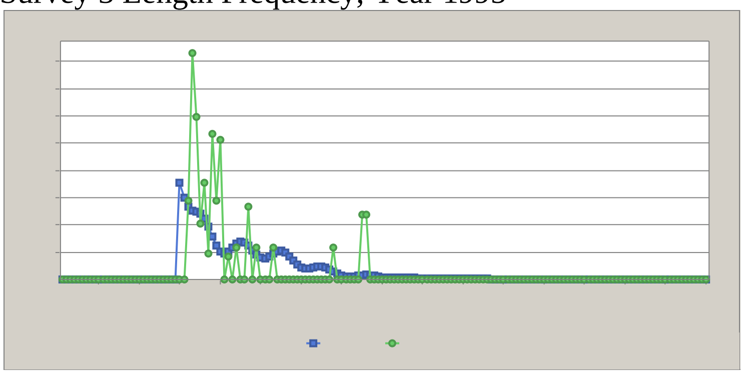

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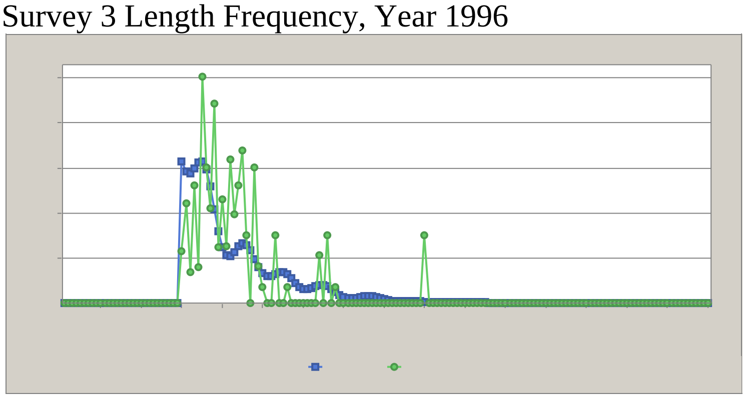

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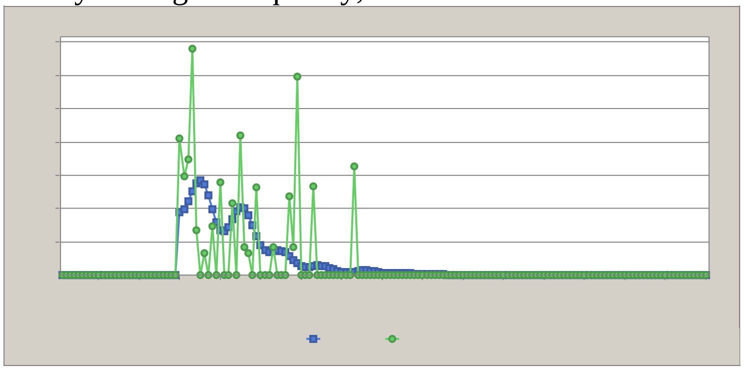
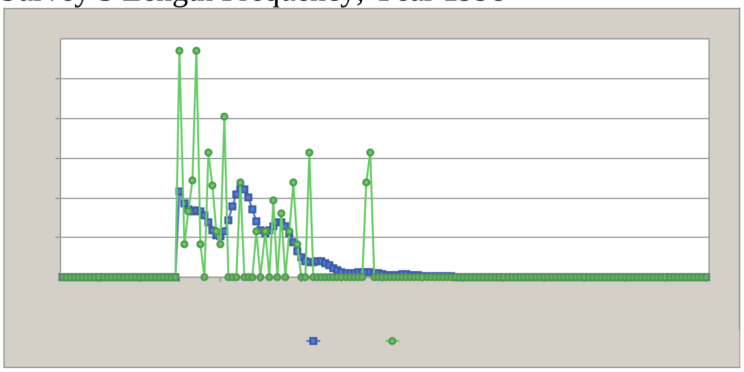
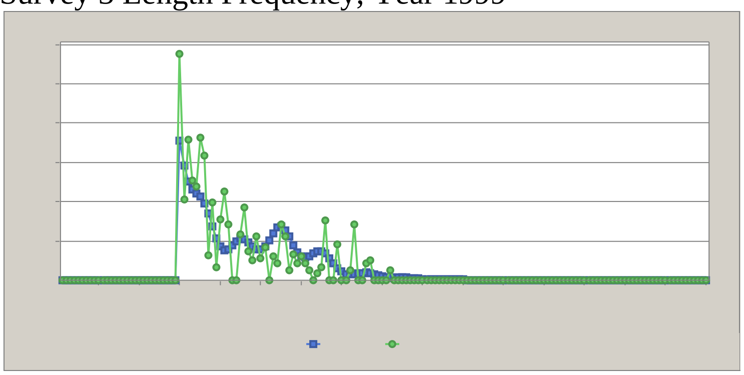


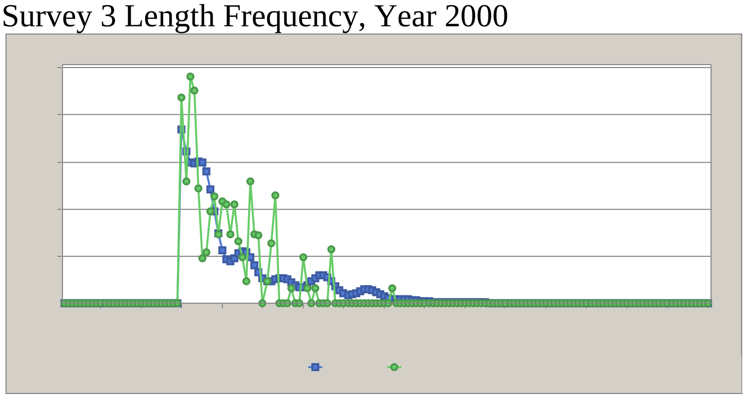
Survey 3 Length Frequency, Year 1998



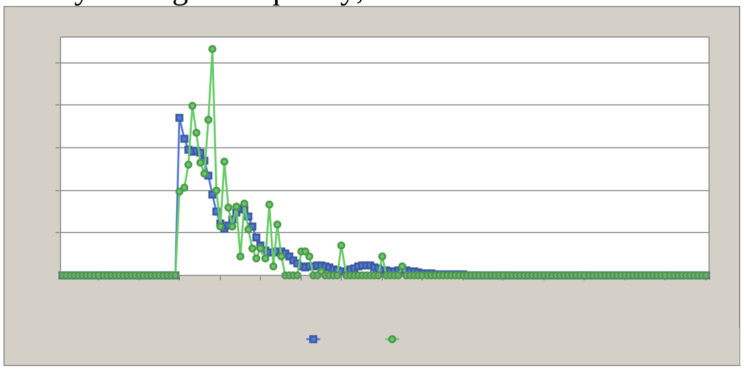
Survey 3 Length Frequency, Year 1999



Survey 3 Length Frequency, Year 2000



Survey 3 Length Frequency, Year 2001



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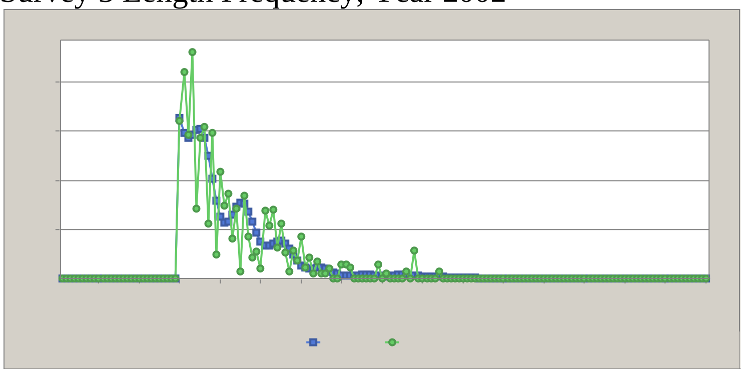
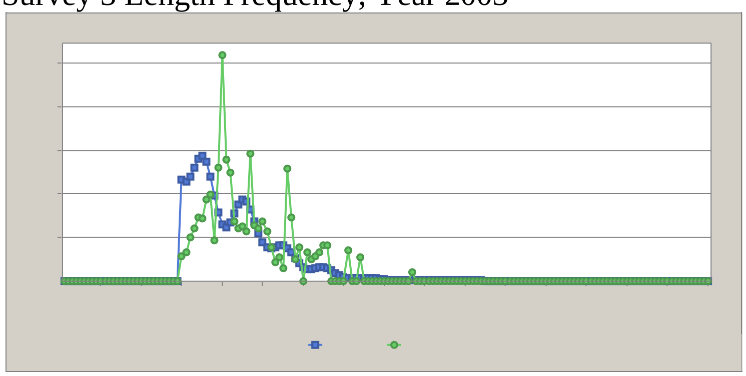
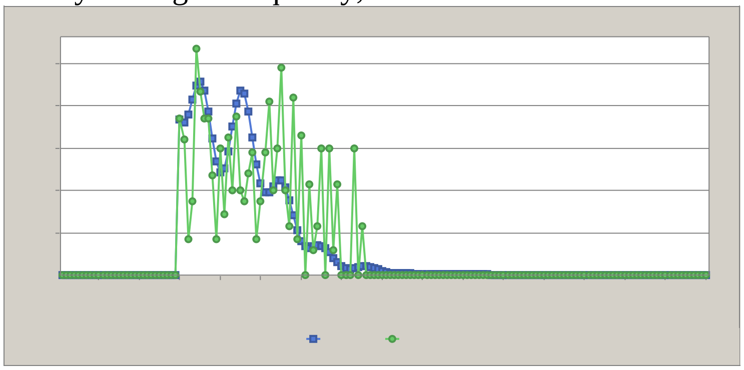


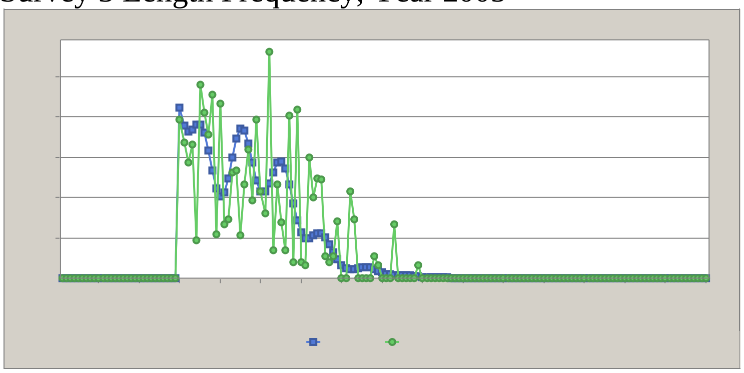
Survey 3 Length Frequency, Year 2003



Survey 3 Length Frequency, Year 2004



Survey 3 Length Frequency, Year 2005



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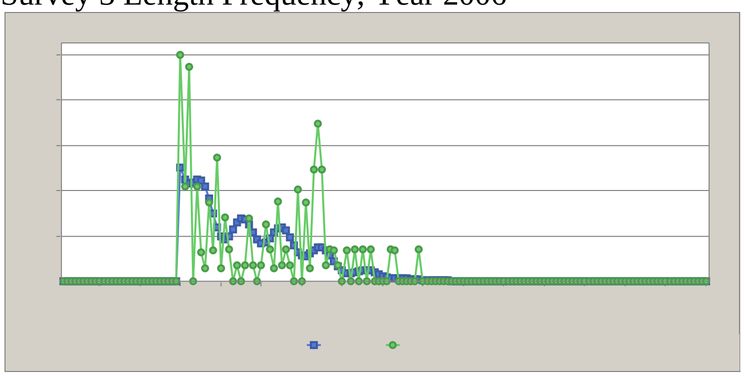
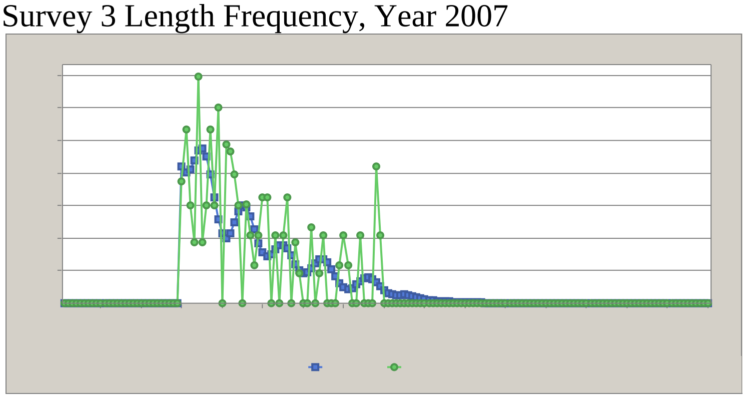
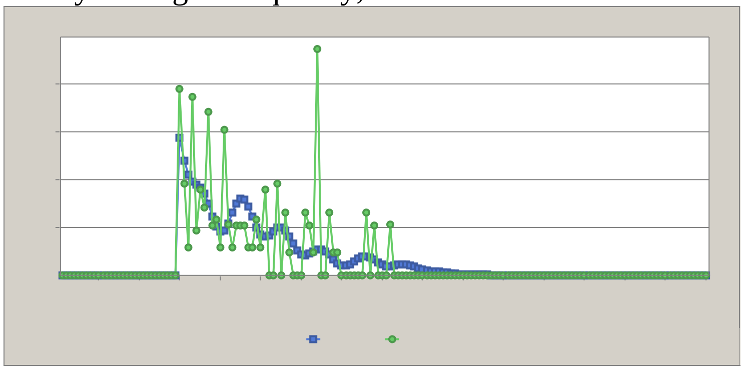


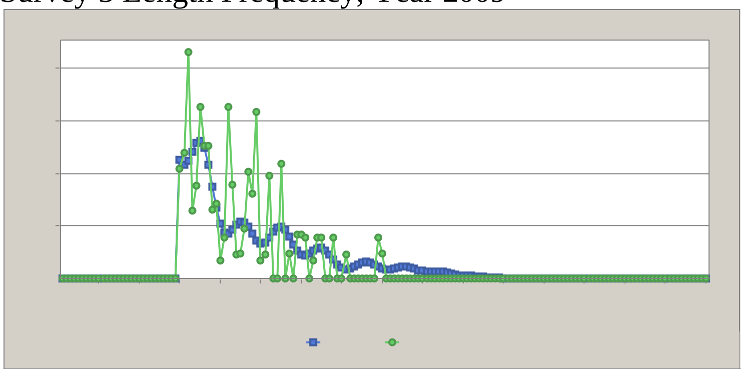
Survey 3 Length Frequency, Year 2007



Survey 3 Length Frequency, Year 2008



Survey 3 Length Frequency, Year 2009



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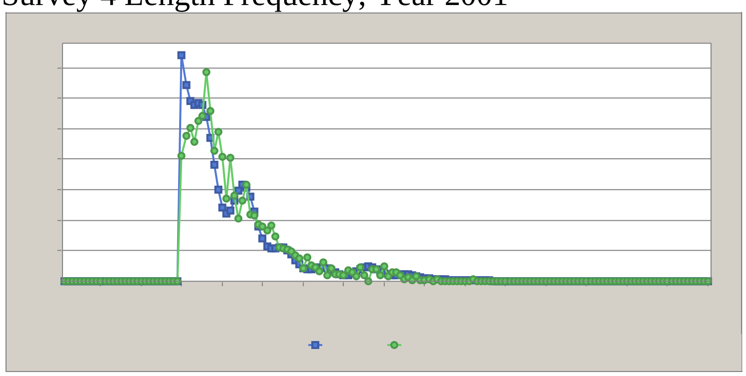

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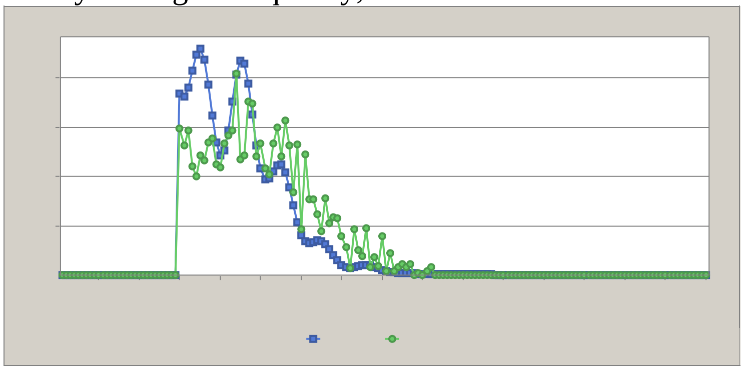
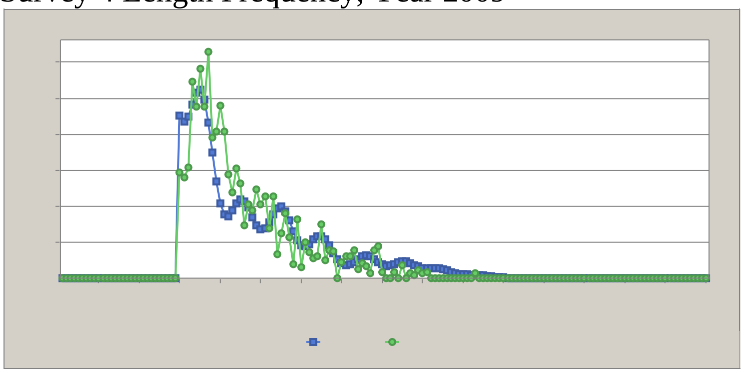
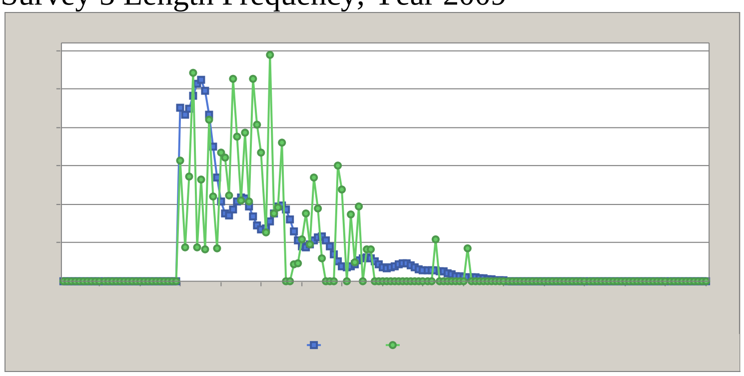


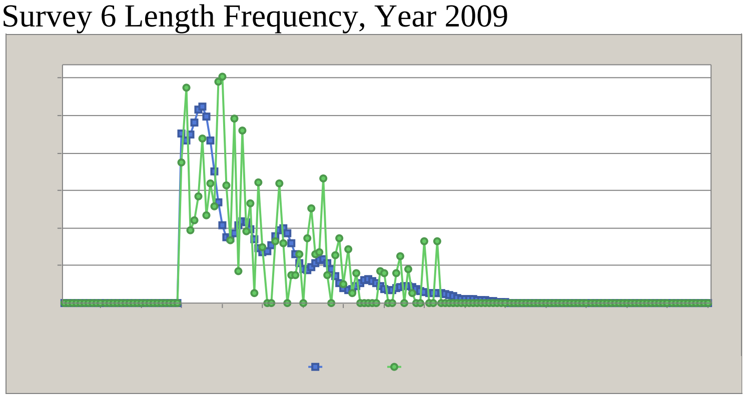
Survey 4 Length Frequency, Year 2009



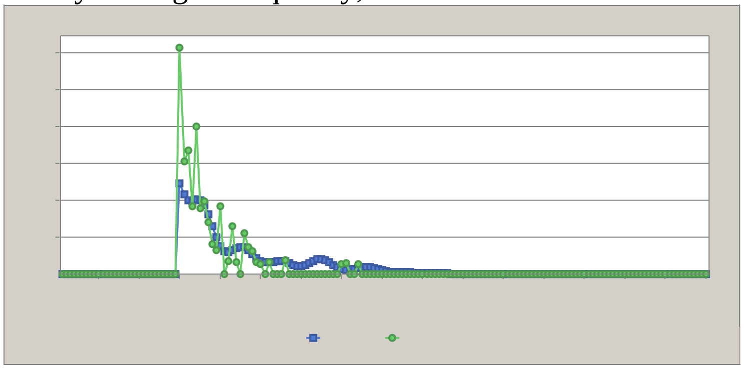
Survey 5 Length Frequency, Year 2009



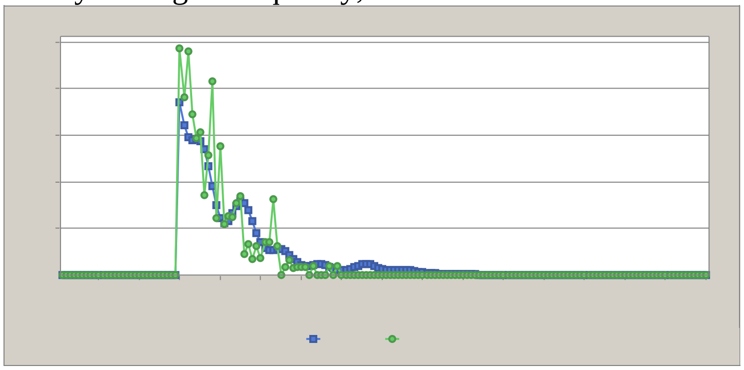
Survey 6 Length Frequency, Year 2009



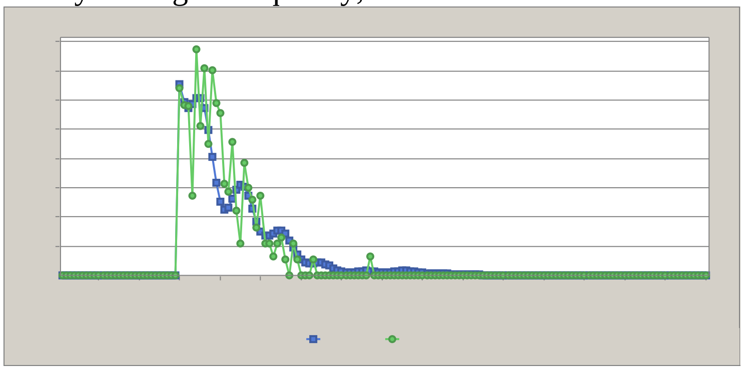
Survey 7 Length Frequency, Year 2000



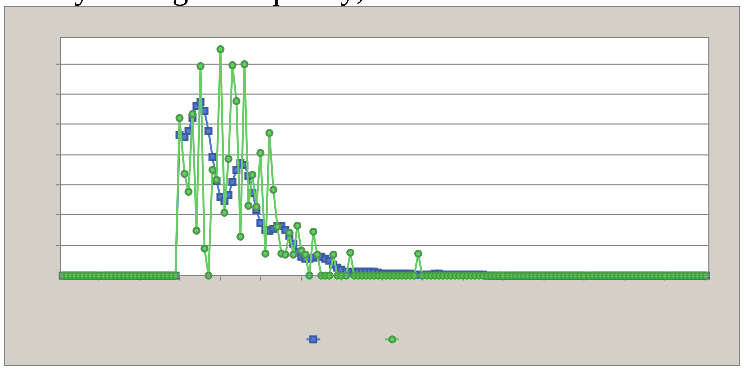
Survey 7 Length Frequency, Year 2001



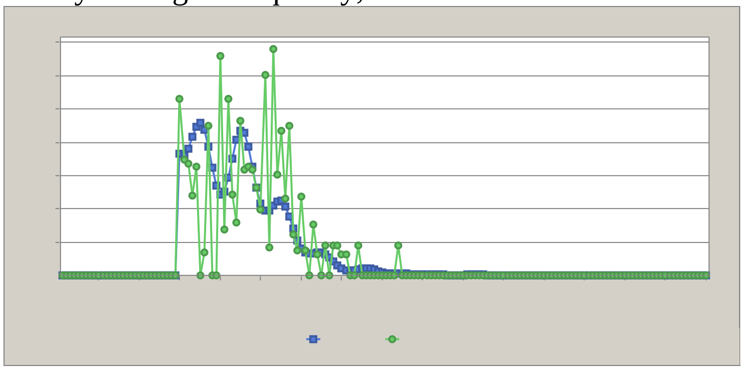
Survey 7 Length Frequency, Year 2002



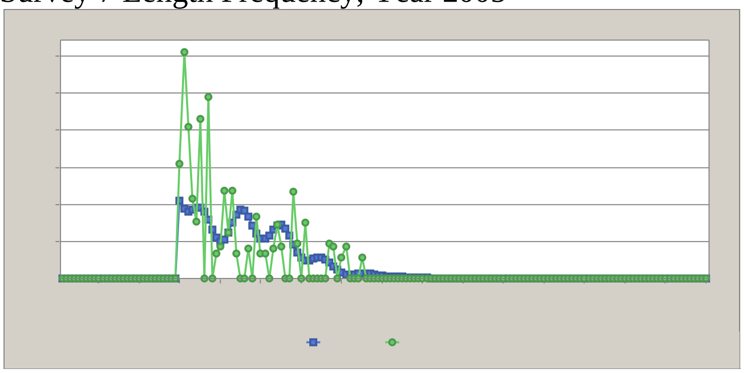
Survey 7 Length Frequency, Year 2003



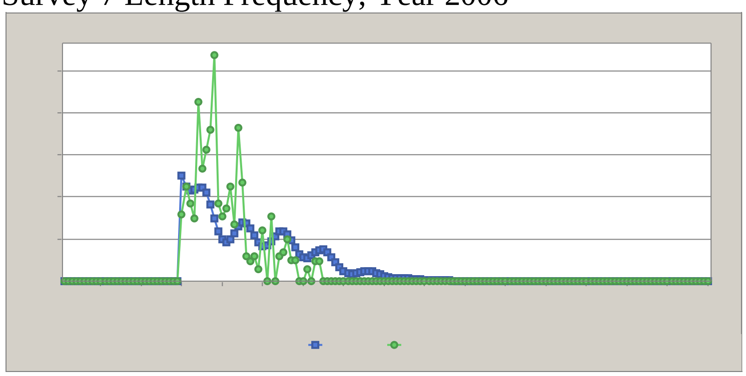
Survey 7 Length Frequency, Year 2004



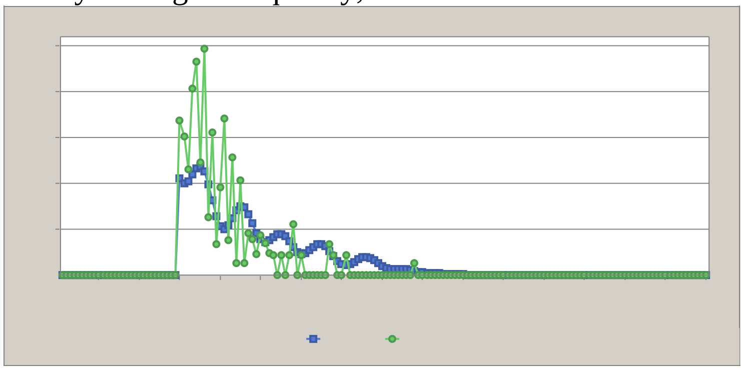
Survey 7 Length Frequency, Year 2005



Survey 7 Length Frequency, Year 2006



Survey 7 Length Frequency, Year 2007



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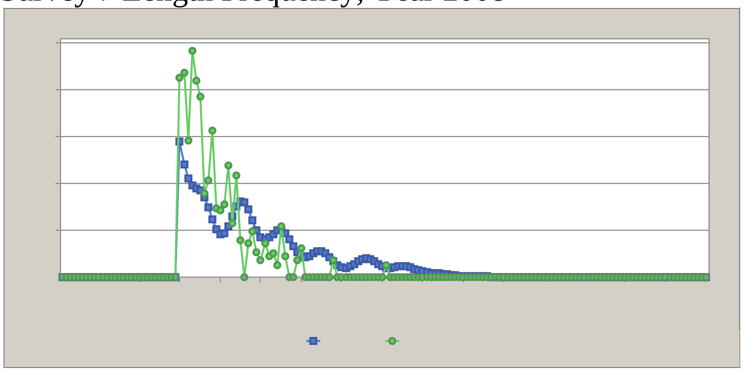
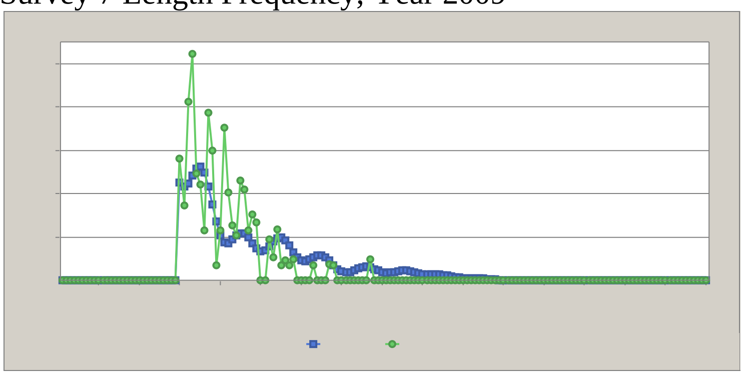
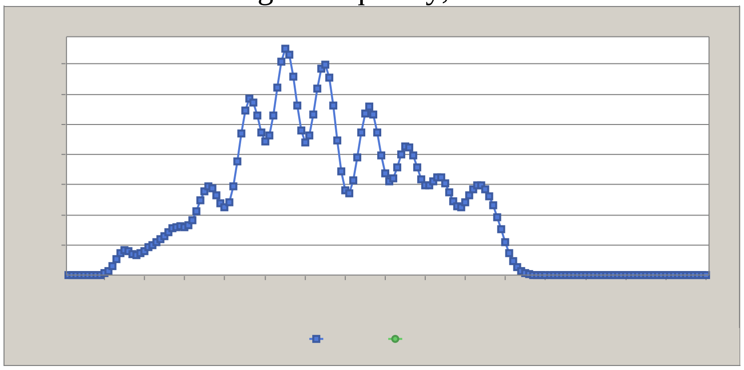


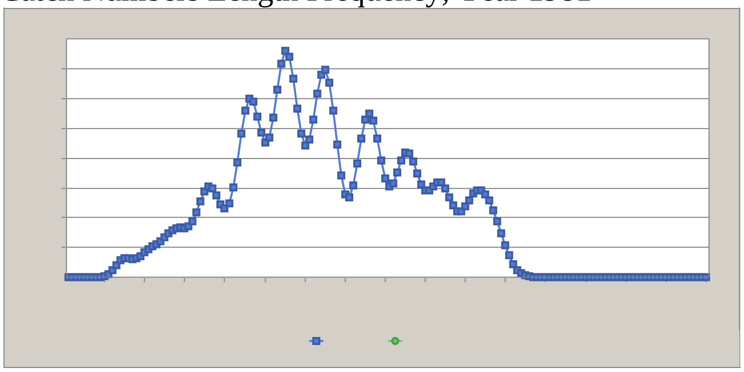
Survey 7 Length Frequency, Year 2009



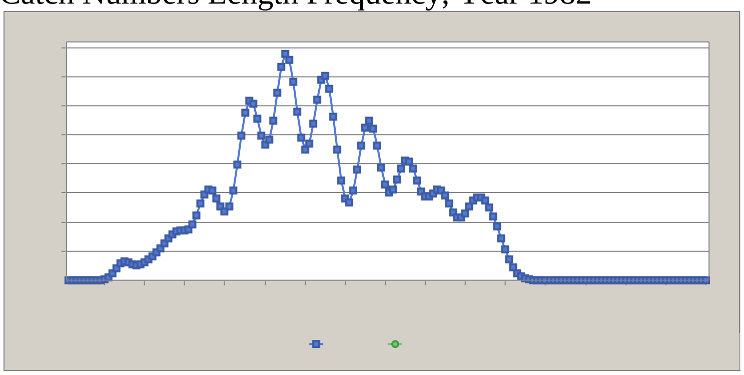
Catch Numbers Length Frequency, Year 1980



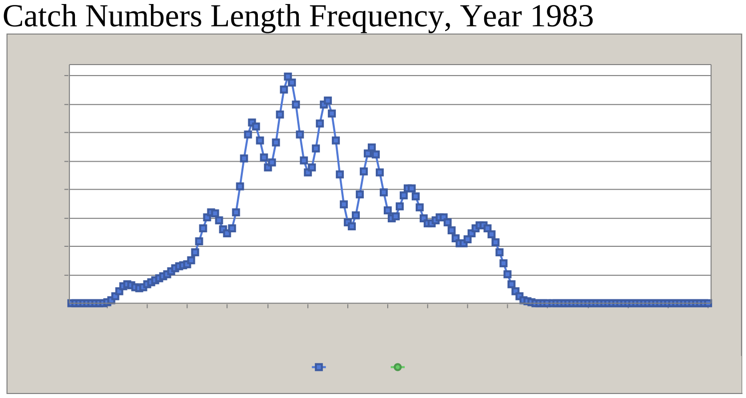
Catch Numbers Length Frequency, Year 1981



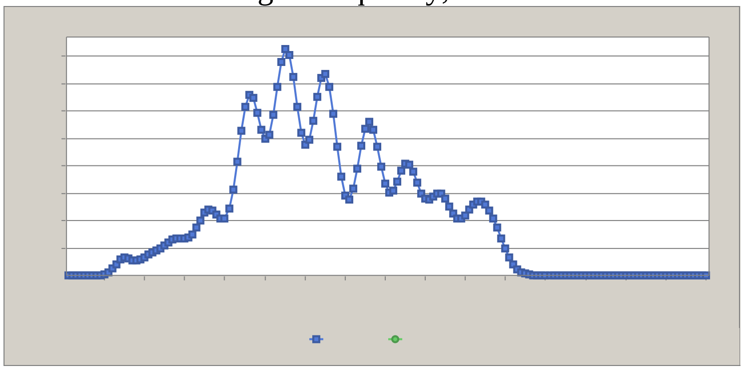
Catch Numbers Length Frequency, Year 1982



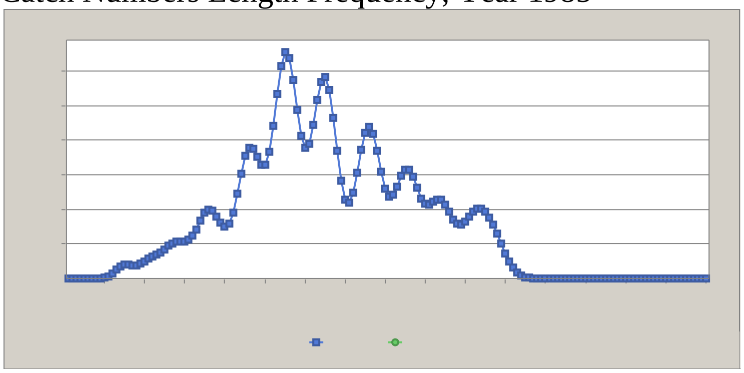
Catch Numbers Length Frequency, Year 1983



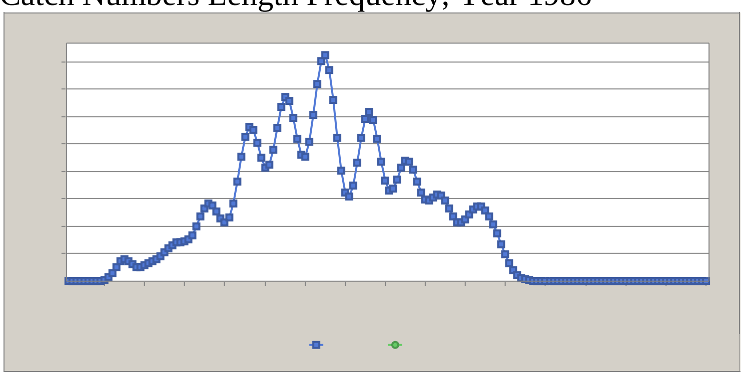
Catch Numbers Length Frequency, Year 1984



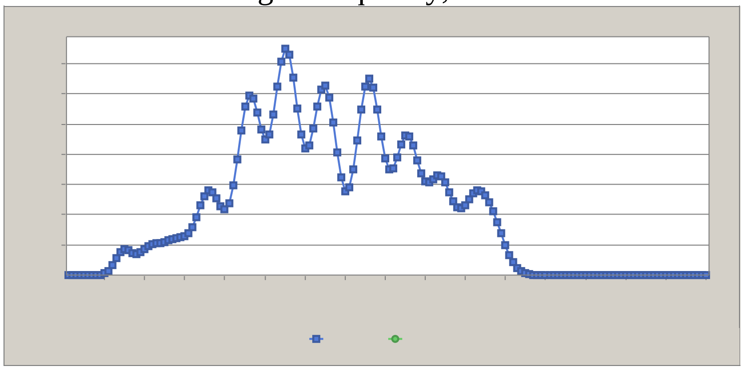
Catch Numbers Length Frequency, Year 1985



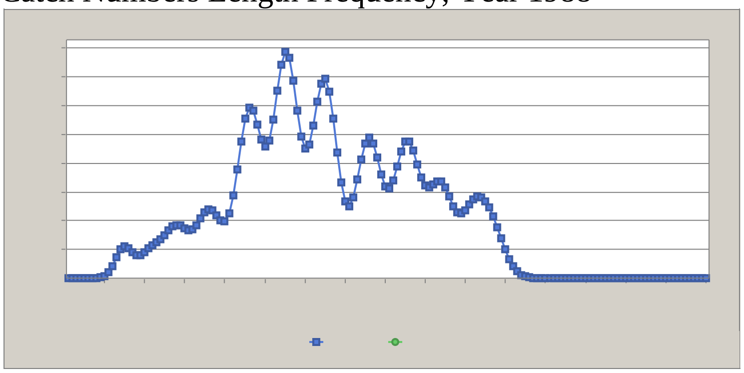
Catch Numbers Length Frequency, Year 1986



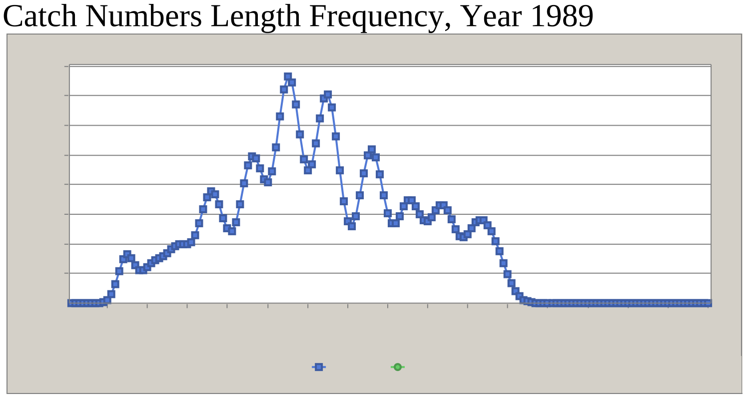
Catch Numbers Length Frequency, Year 1987



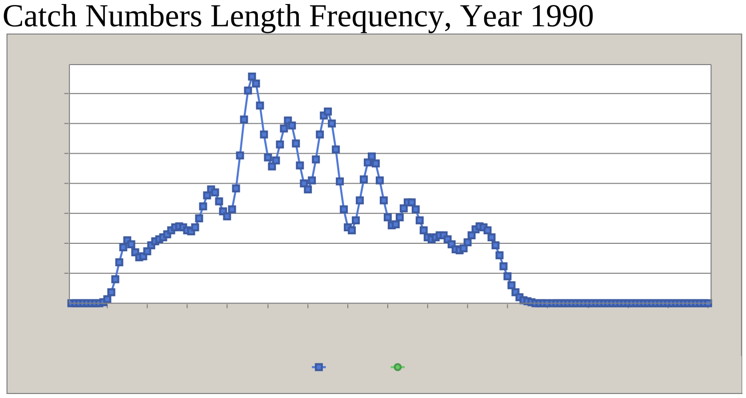
Catch Numbers Length Frequency, Year 1988



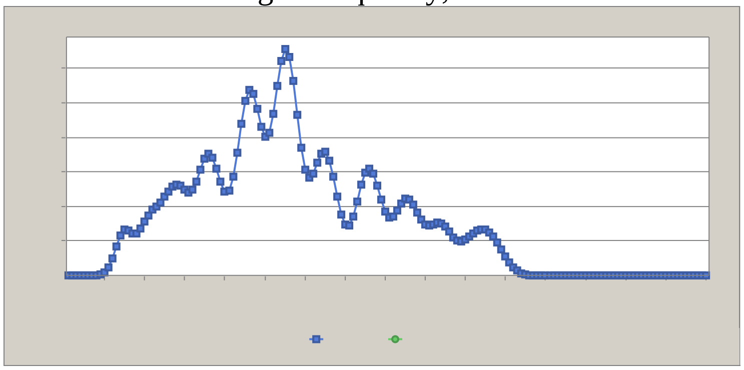
Catch Numbers Length Frequency, Year 1989



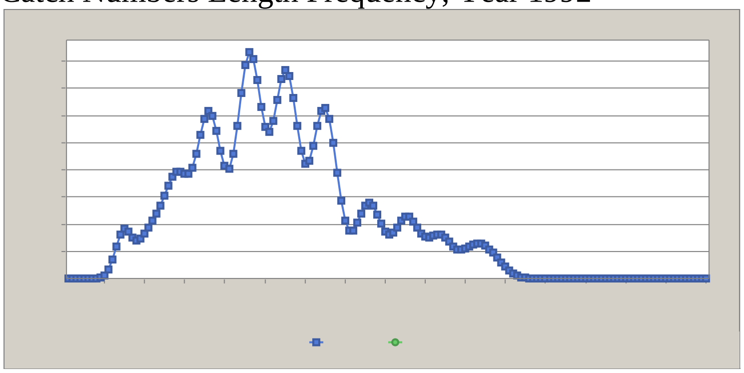
Catch Numbers Length Frequency, Year 1990



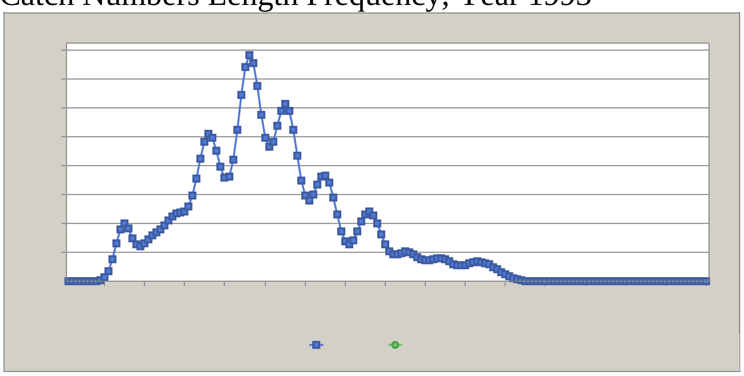
Catch Numbers Length Frequency, Year 1991



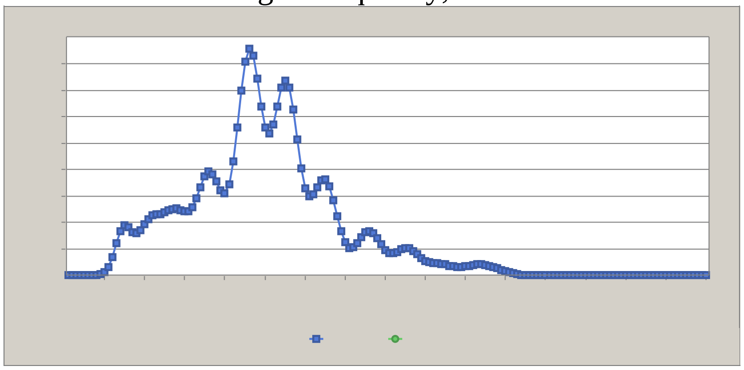
Catch Numbers Length Frequency, Year 1992



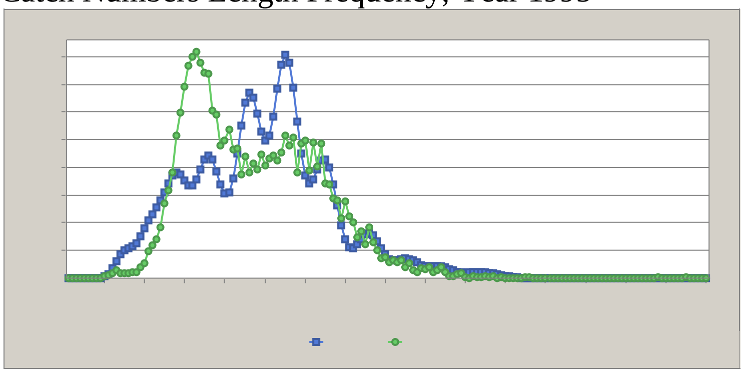
Catch Numbers Length Frequency, Year 1993



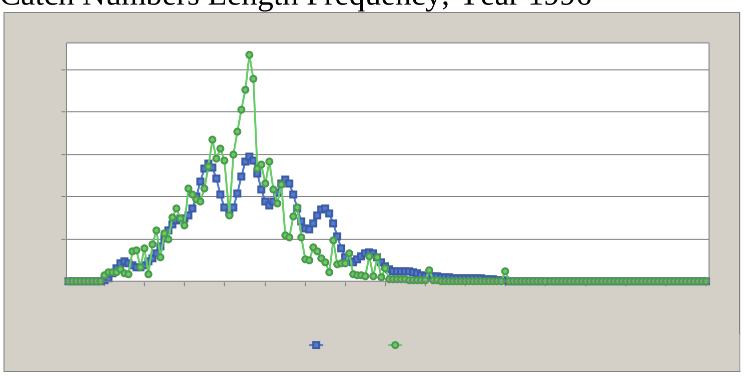
Catch Numbers Length Frequency, Year 1994



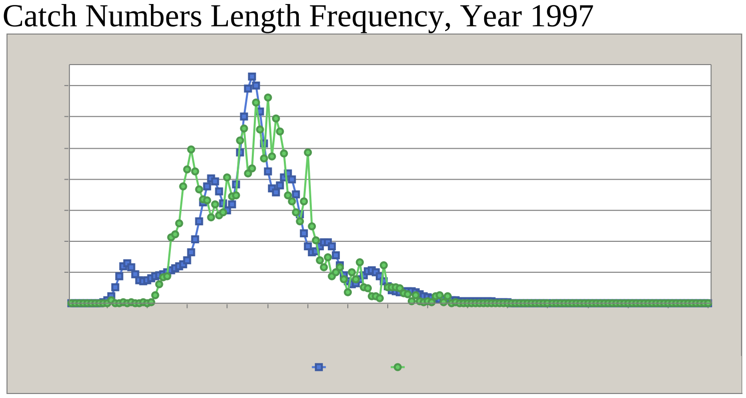
Catch Numbers Length Frequency, Year 1995



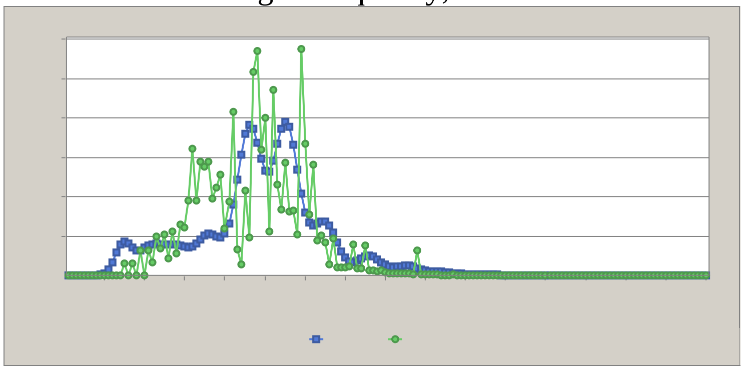
Catch Numbers Length Frequency, Year 1996



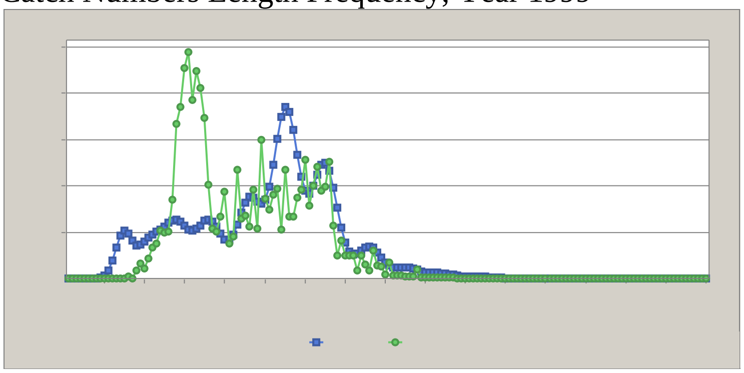
Catch Numbers Length Frequency, Year 1997



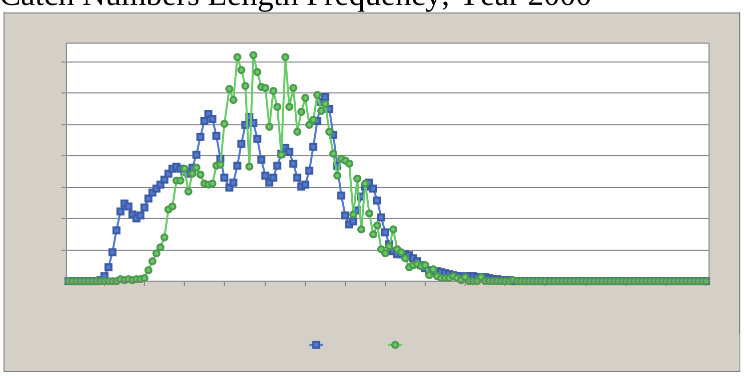
Catch Numbers Length Frequency, Year 1998



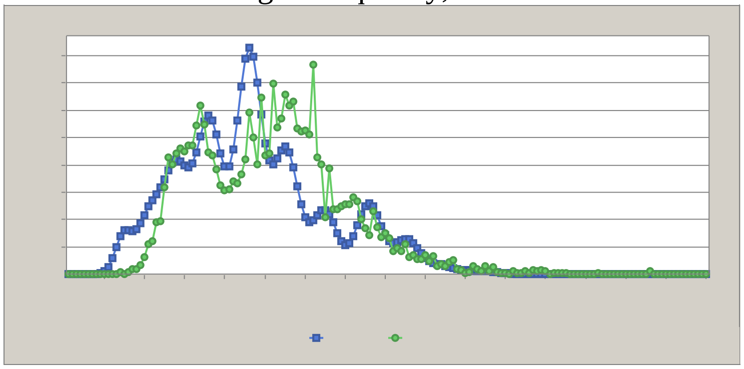
Catch Numbers Length Frequency, Year 1999



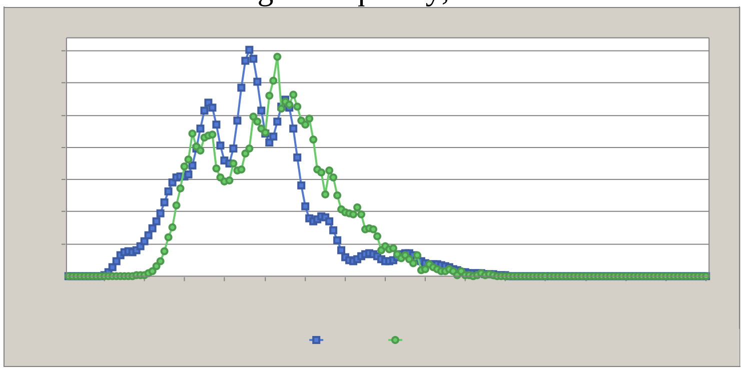
Catch Numbers Length Frequency, Year 2000



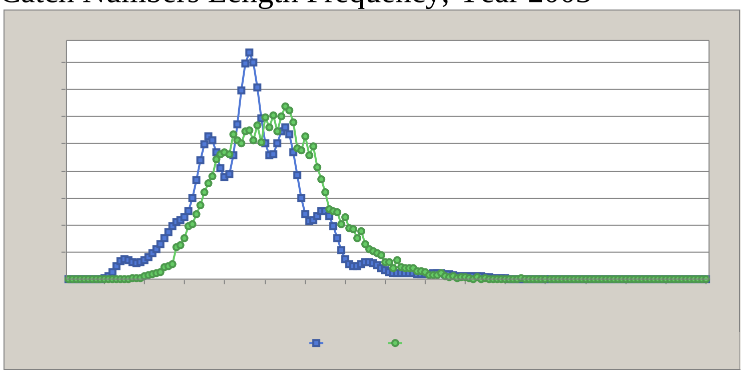
Catch Numbers Length Frequency, Year 2001



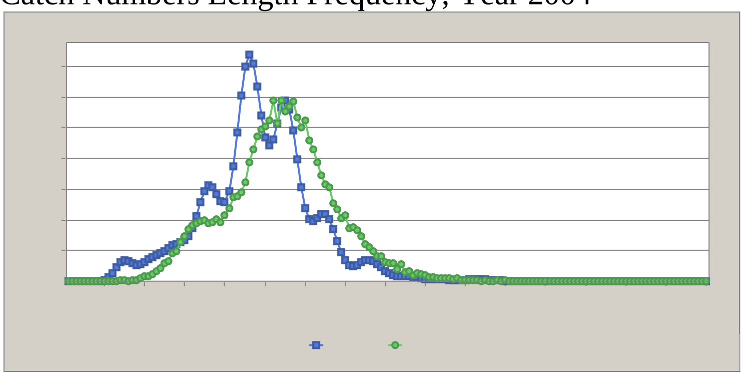
Catch Numbers Length Frequency, Year 2002



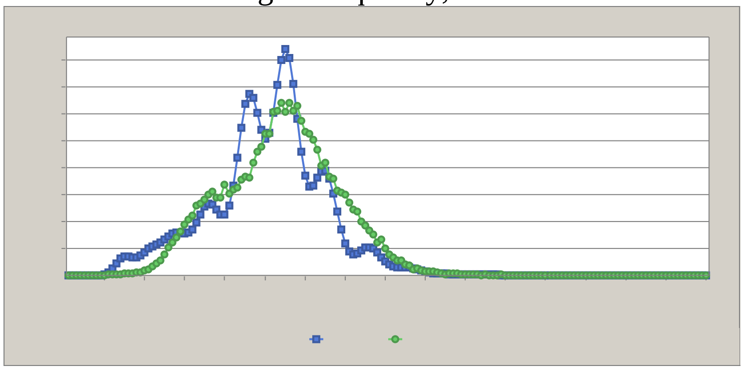
Catch Numbers Length Frequency, Year 2003



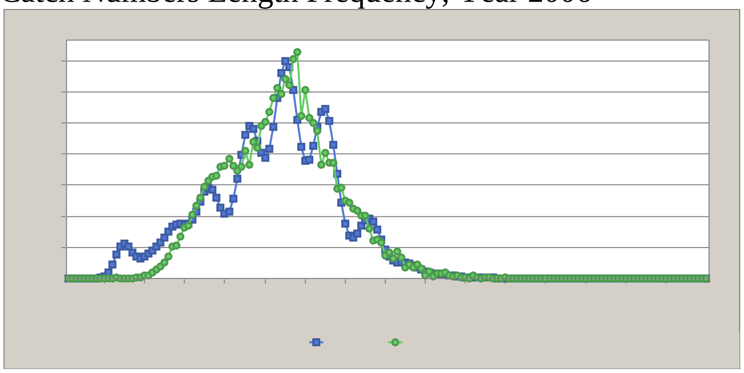
Catch Numbers Length Frequency, Year 2004



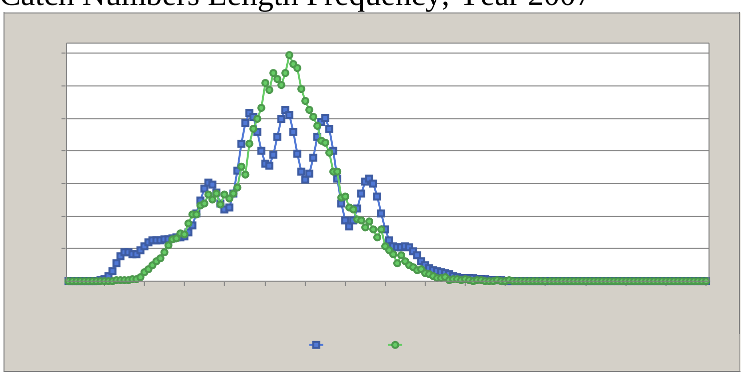
Catch Numbers Length Frequency, Year 2005



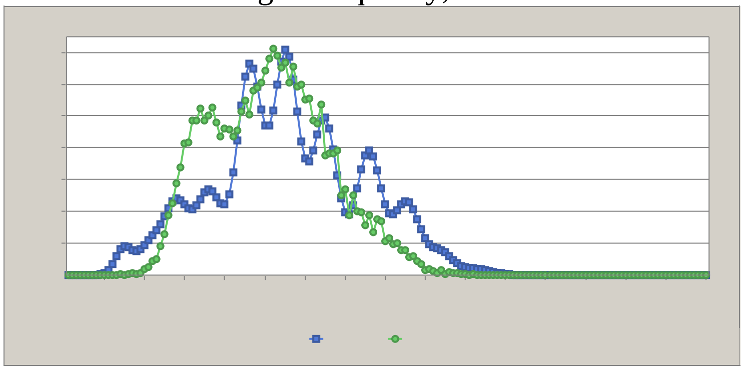
Catch Numbers Length Frequency, Year 2006



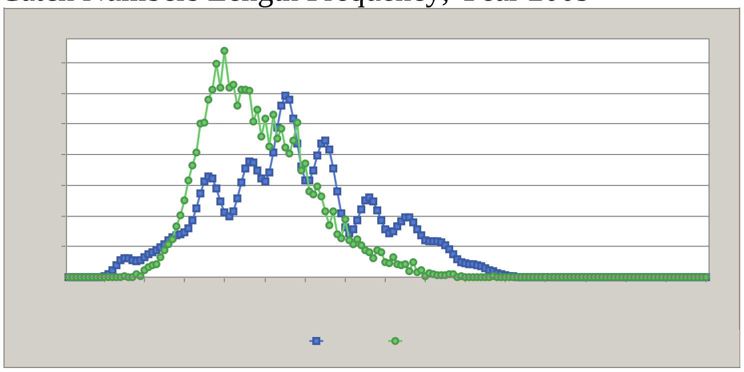
Catch Numbers Length Frequency, Year 2007



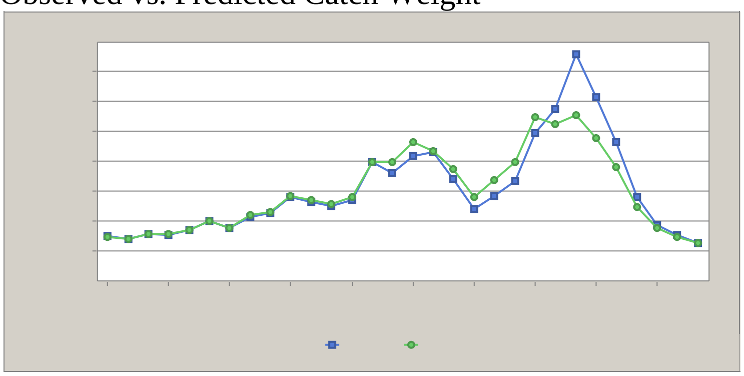
Catch Numbers Length Frequency, Year 2008



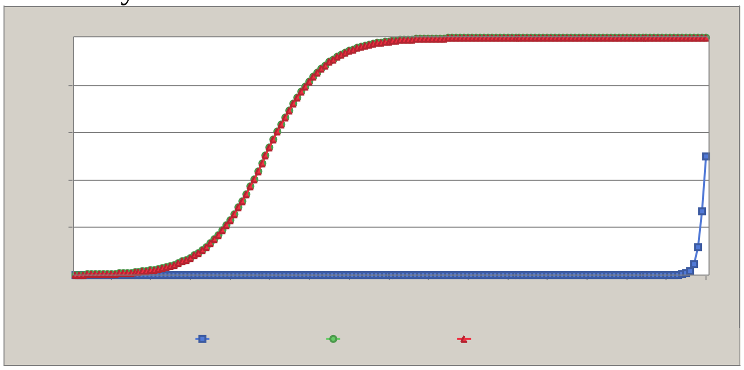
Catch Numbers Length Frequency, Year 2009



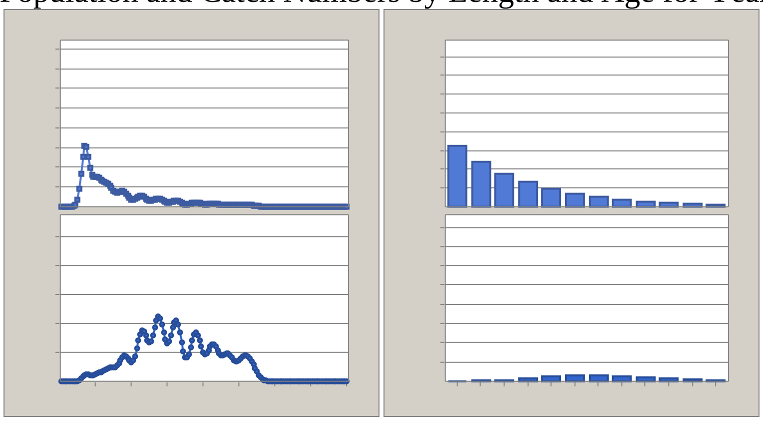
Observed vs. Predicted Catch Weight



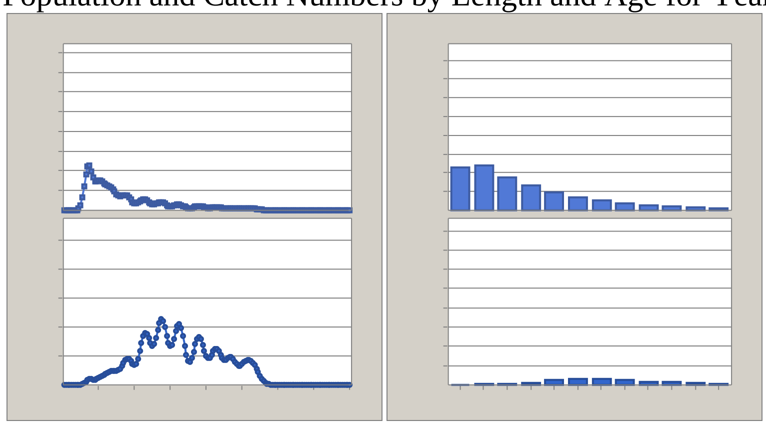
Selectivity for Block 1



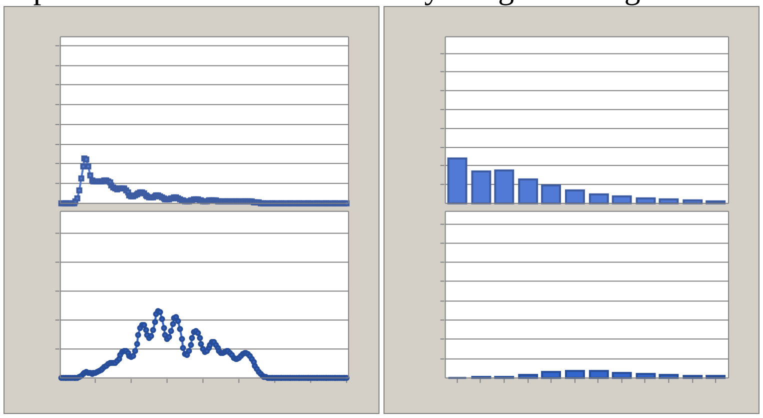
Population and Catch Numbers by Length and Age for Year 1980



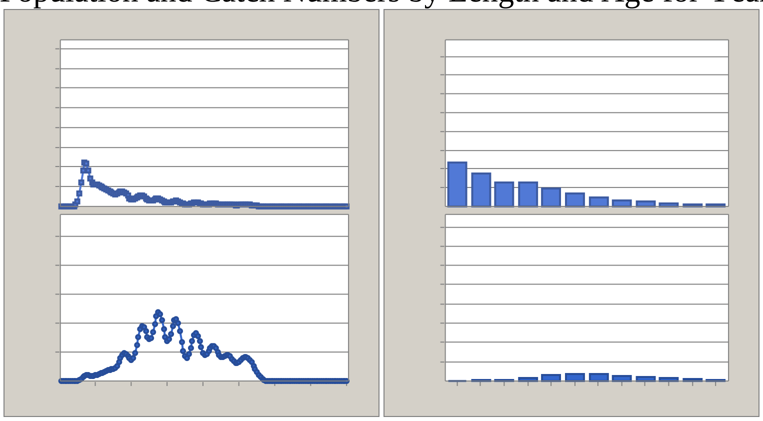
Population and Catch Numbers by Length and Age for Year 1981



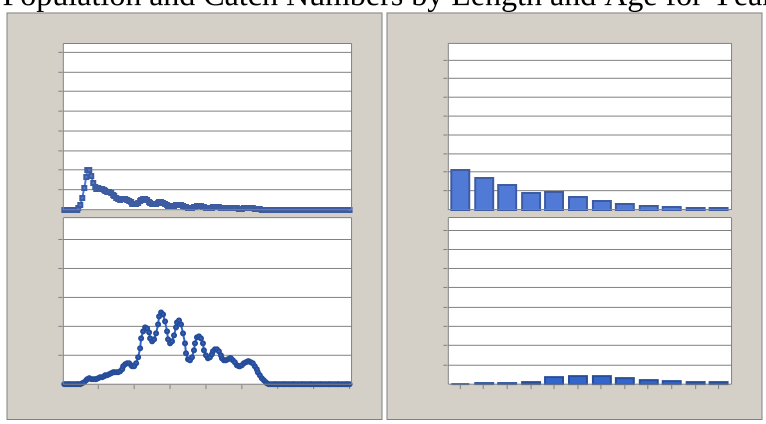
Population and Catch Numbers by Length and Age for Year 1982



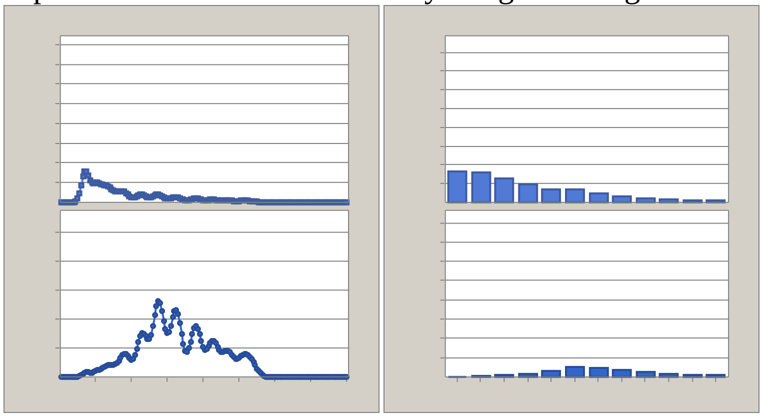
Population and Catch Numbers by Length and Age for Year 1983



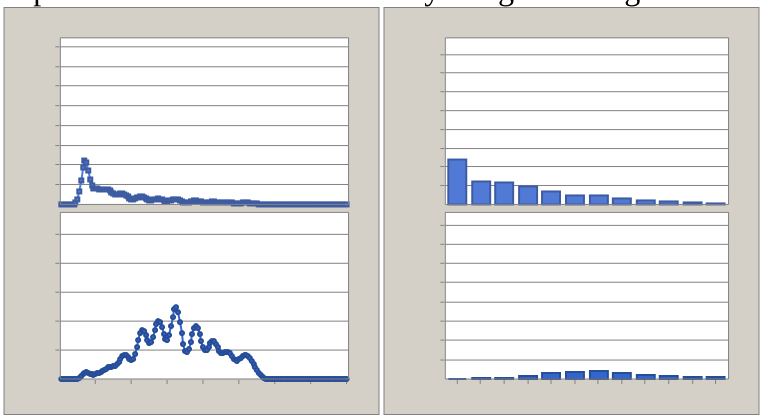
Population and Catch Numbers by Length and Age for Year 1984



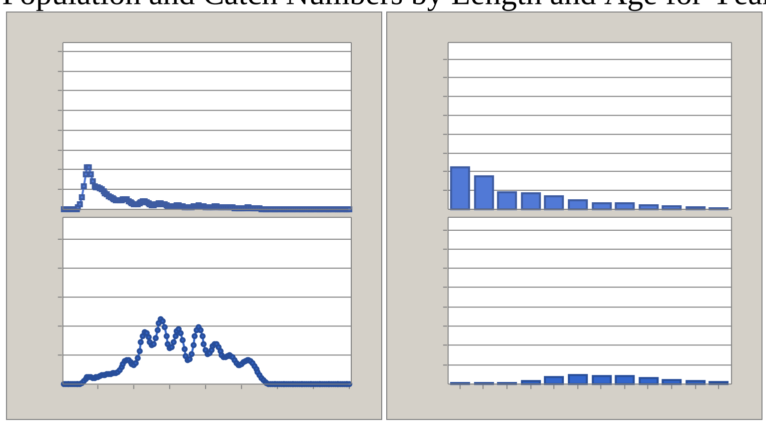
Population and Catch Numbers by Length and Age for Year 1985



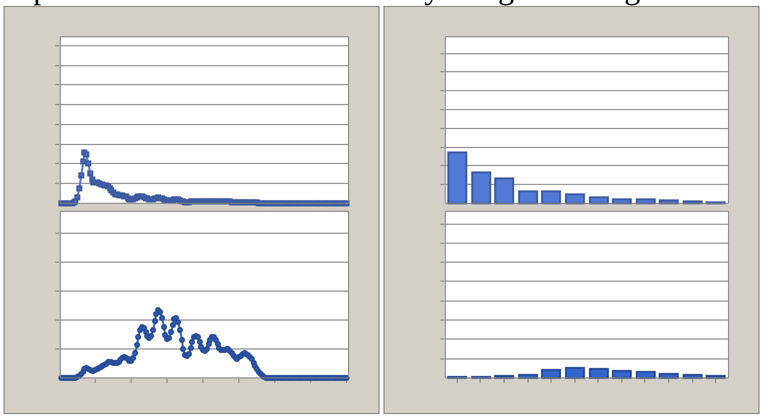
Population and Catch Numbers by Length and Age for Year 1986



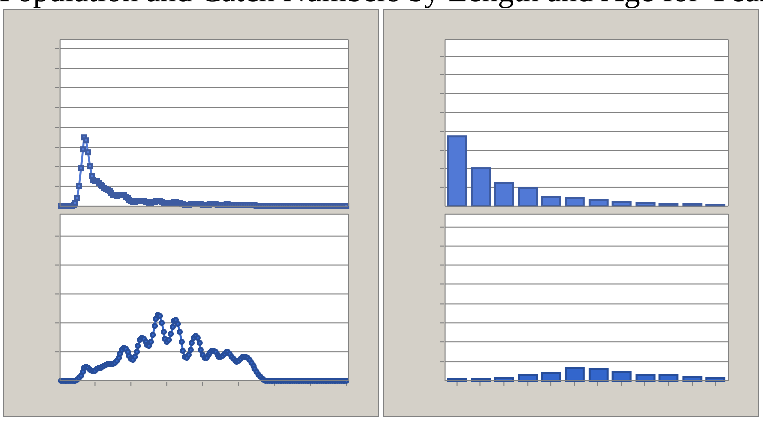
Population and Catch Numbers by Length and Age for Year 1987



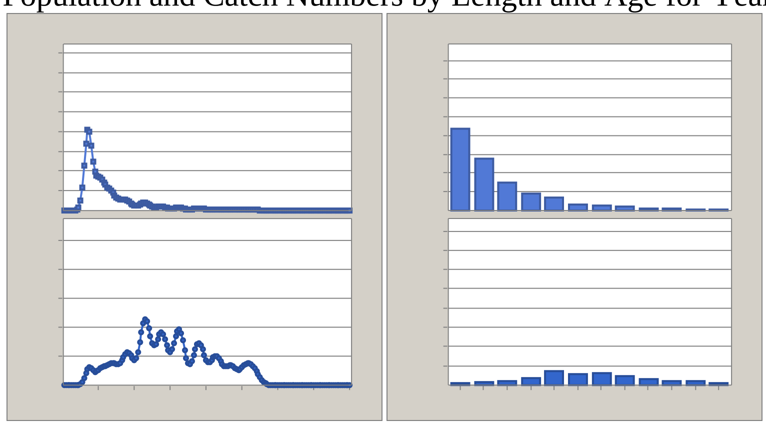
Population and Catch Numbers by Length and Age for Year 1988



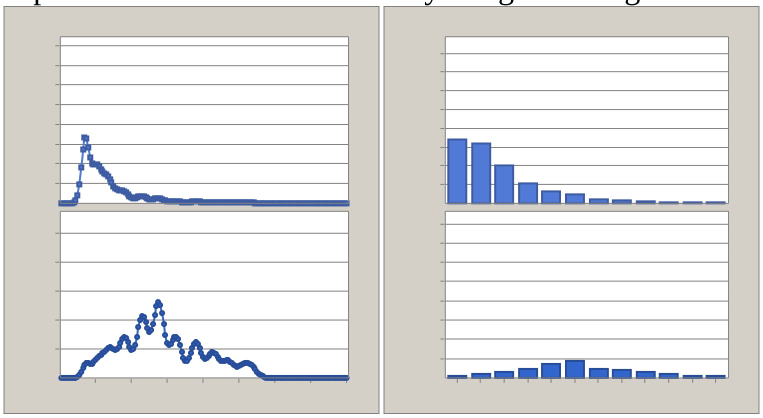
Population and Catch Numbers by Length and Age for Year 1989



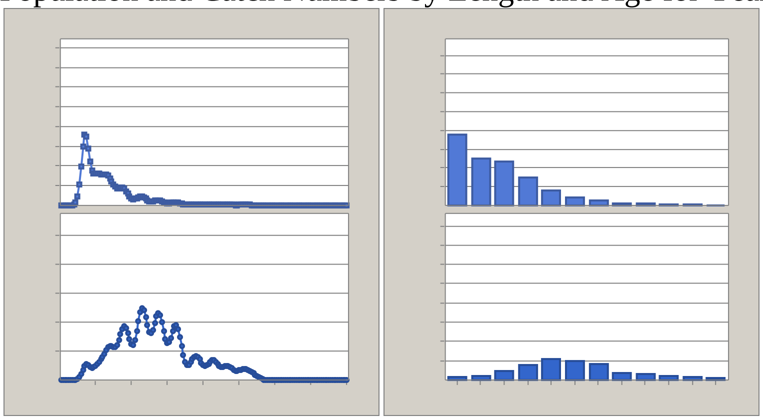
Population and Catch Numbers by Length and Age for Year 1990



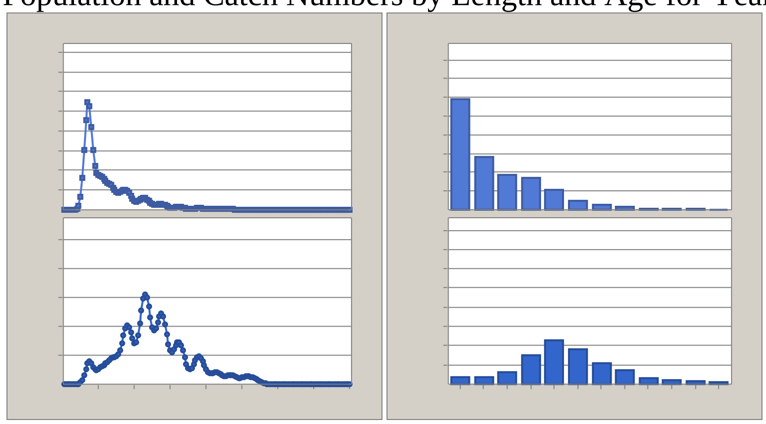
Population and Catch Numbers by Length and Age for Year 1991



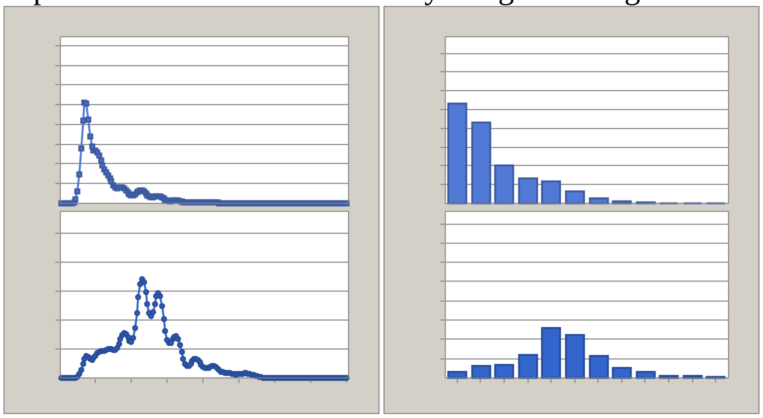
Population and Catch Numbers by Length and Age for Year 1992



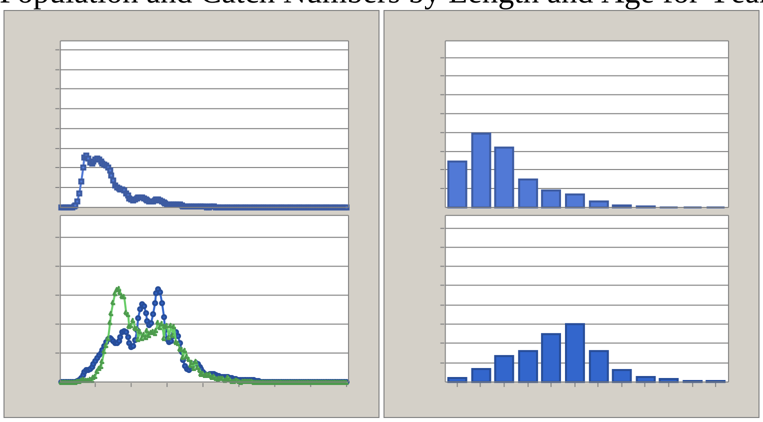
Population and Catch Numbers by Length and Age for Year 1993



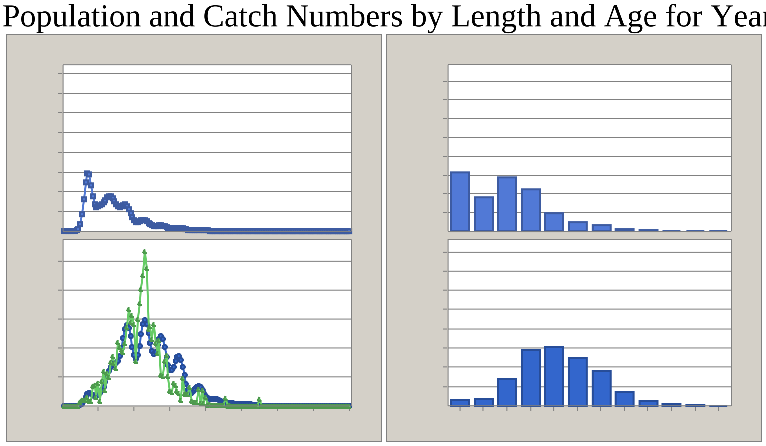
Population and Catch Numbers by Length and Age for Year 1994



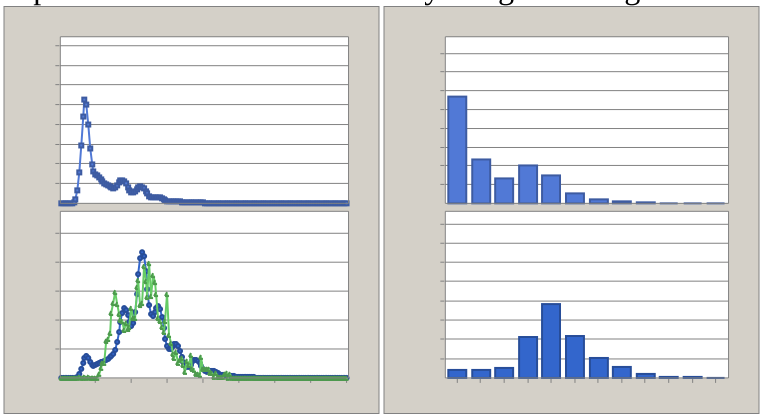
Population and Catch Numbers by Length and Age for Year 1995



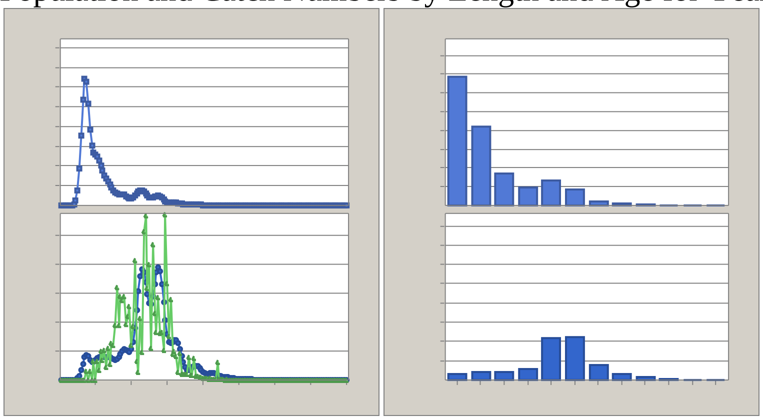
Population and Catch Numbers by Length and Age for Year 1996



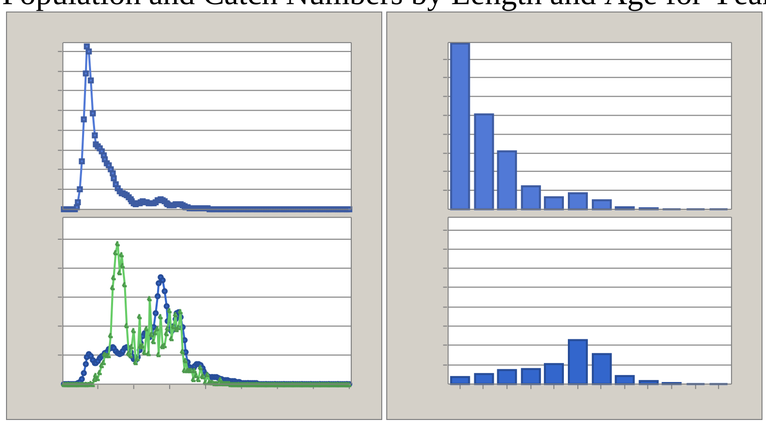
Population and Catch Numbers by Length and Age for Year 1997



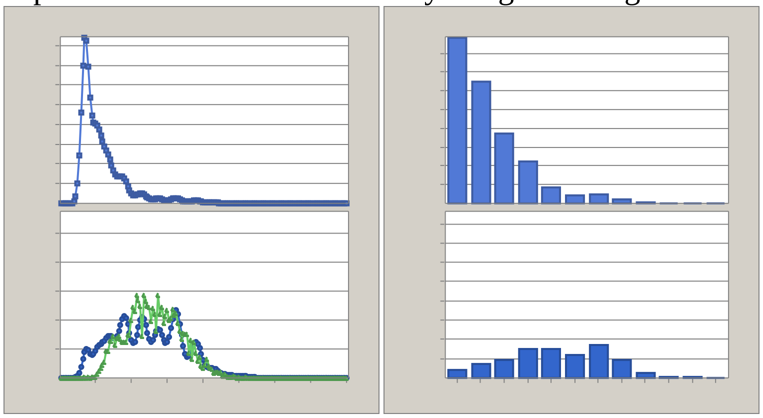
Population and Catch Numbers by Length and Age for Year 1998



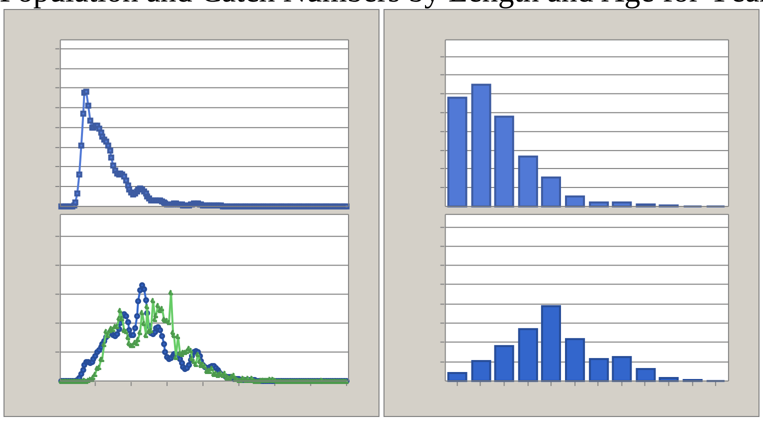
Population and Catch Numbers by Length and Age for Year 1999



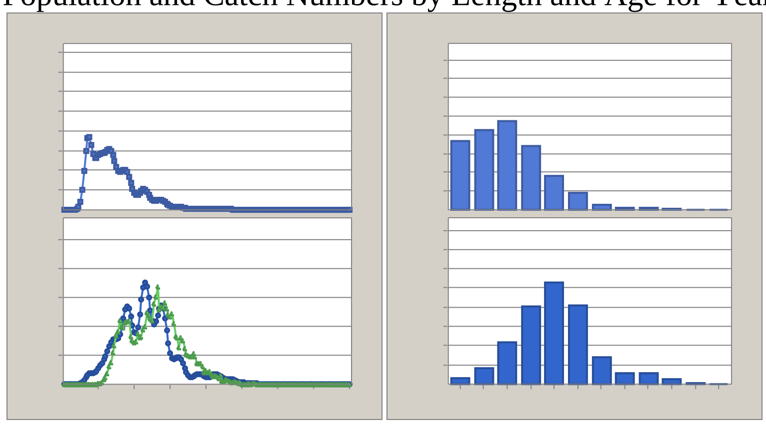
Population and Catch Numbers by Length and Age for Year 2000



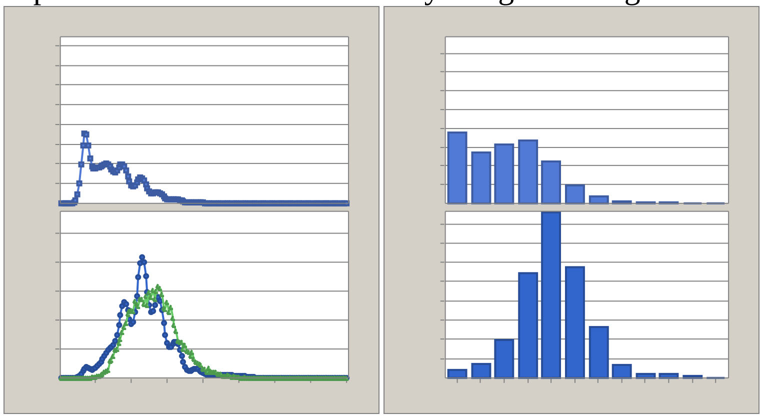
Population and Catch Numbers by Length and Age for Year 2001



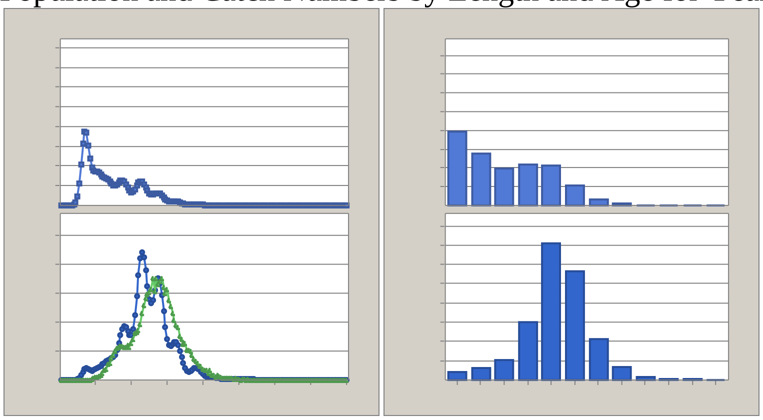
Population and Catch Numbers by Length and Age for Year 2002



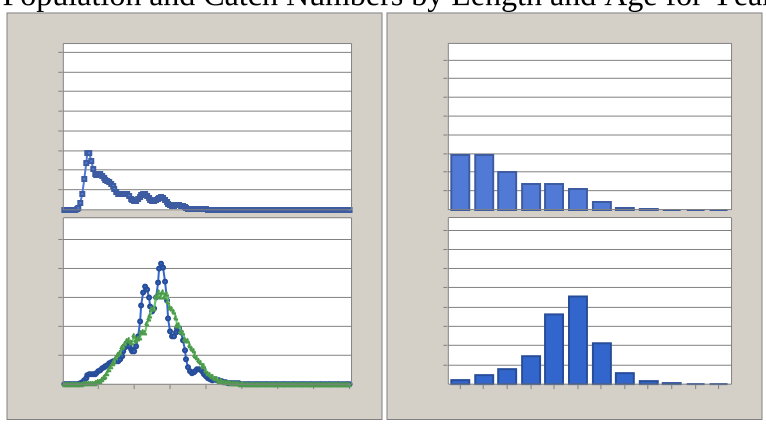
Population and Catch Numbers by Length and Age for Year 2003



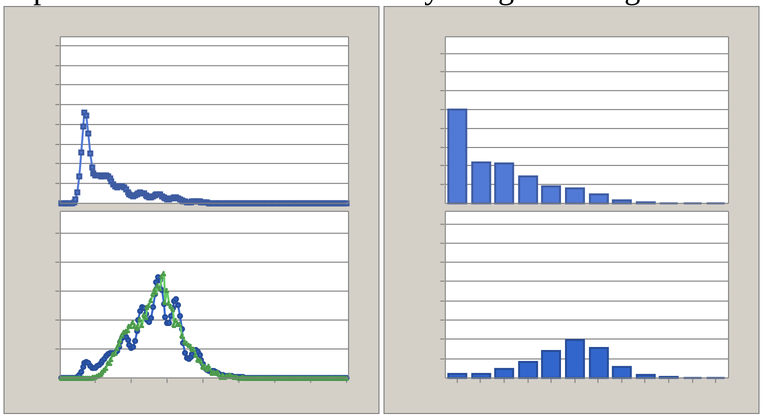
Population and Catch Numbers by Length and Age for Year 2004



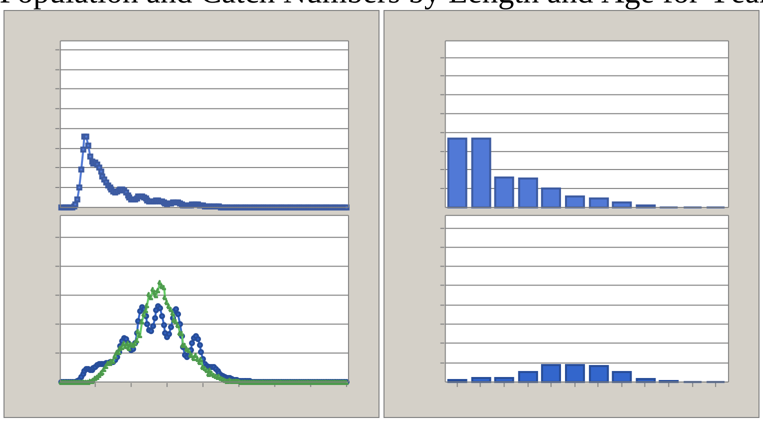
Population and Catch Numbers by Length and Age for Year 2005



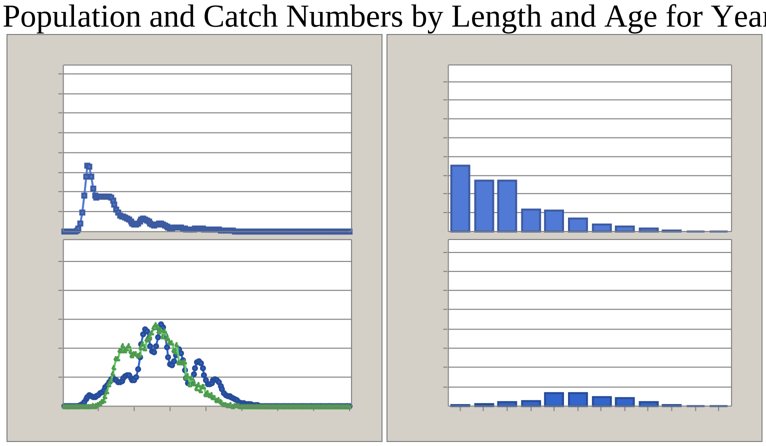
Population and Catch Numbers by Length and Age for Year 2006



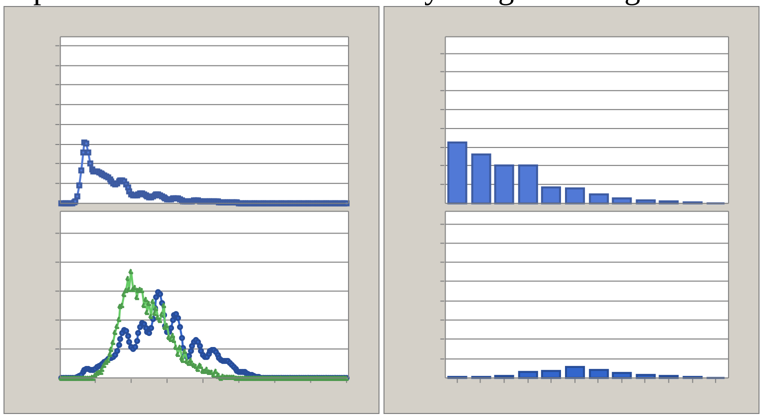
Population and Catch Numbers by Length and Age for Year 2007



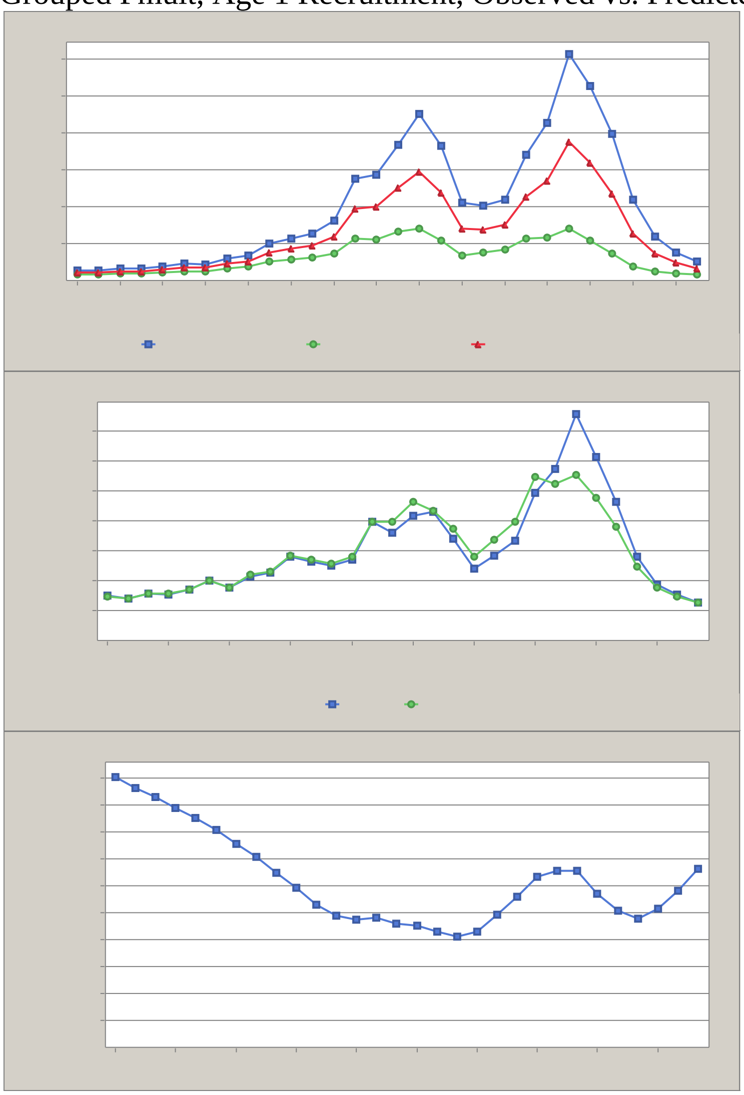
Population and Catch Numbers by Length and Age for Year 2008



Population and Catch Numbers by Length and Age for Year 2009



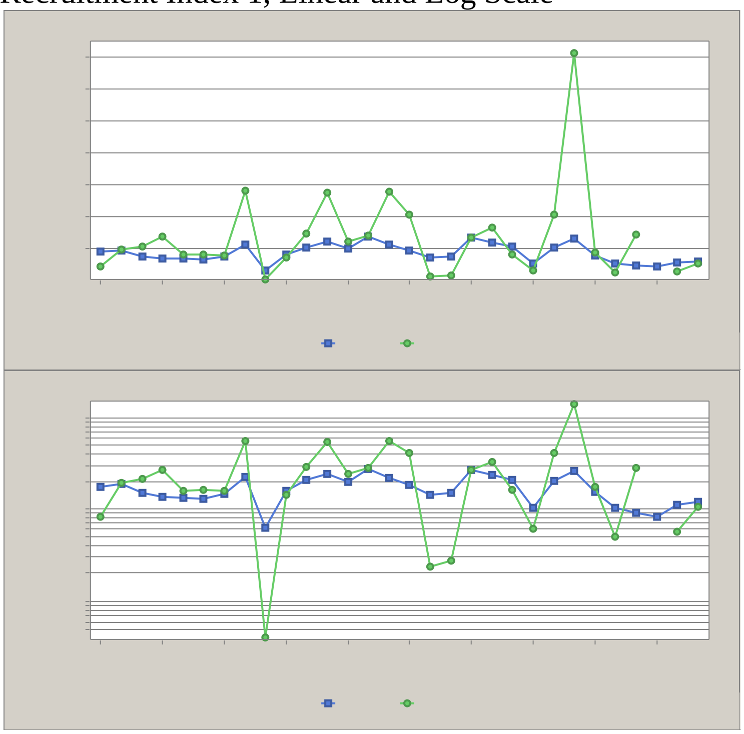
Grouped Fmult, Age 1 Recruitment, Observed vs. Predicted Catch Weight, and Total Biomass



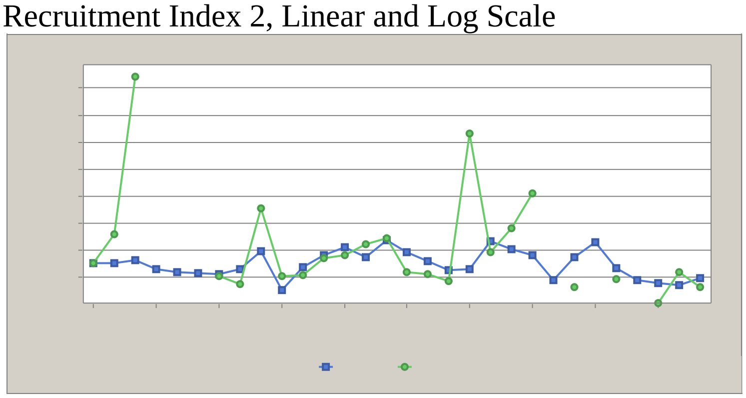
Southern Management Area Final Run 8

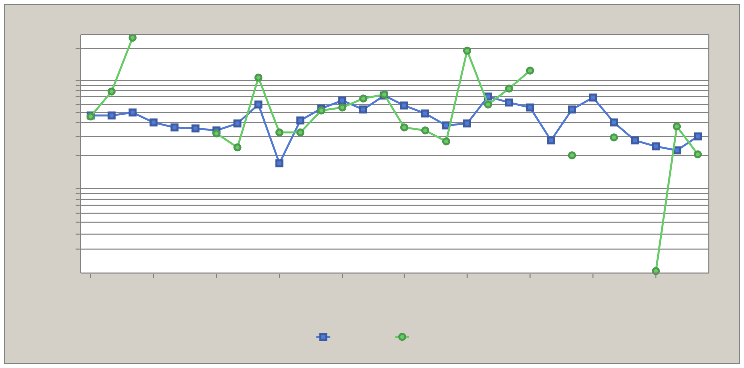
Recruitment Indices, Group Linear and Log Scale, 1 Index per Line (14 Plots)
Adult Indices, Group Linear and Log Scale, 1 Index per Line (8 Plots)
Survey Length Frequencies (210 Plots)
Catch Numbers, Catch Length Frequency (30 Plots)
Observed vs. Predicted Catch Weight (1 Plot)
Selectivity (2 Plots)
4-Plot: Population and Catch Numbers (60 Plots)
Fmult, Age 1 Recruitment, Observed vs. Predicted Catch Weight, and Total Biomass: Group 2 per Line (4 Plots)

Recruitment Index 1, Linear and Log Scale

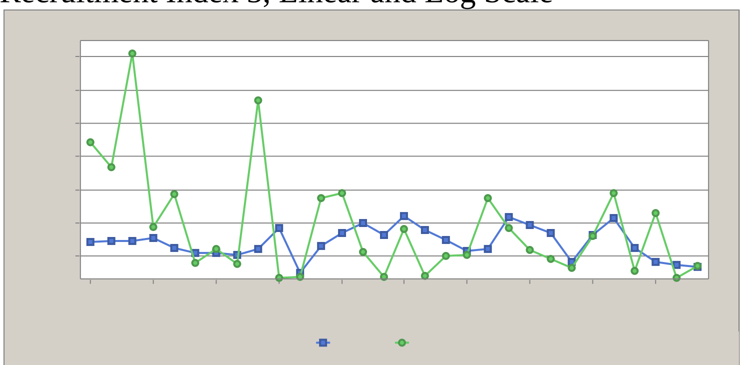


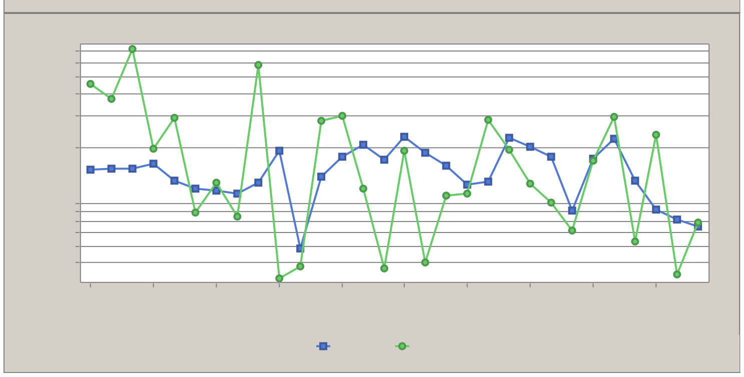
Recruitment Index 2, Linear and Log Scale



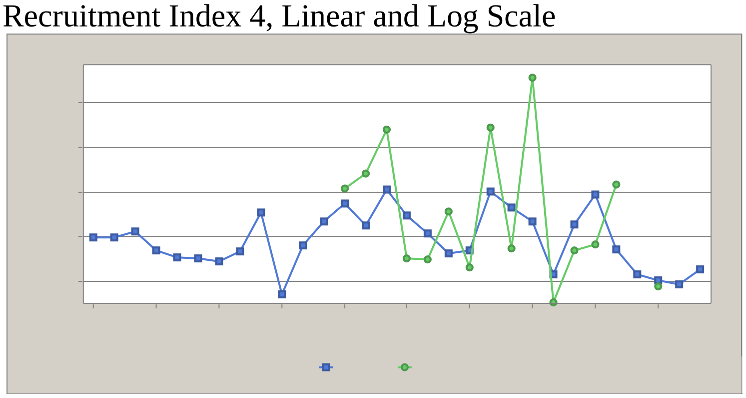


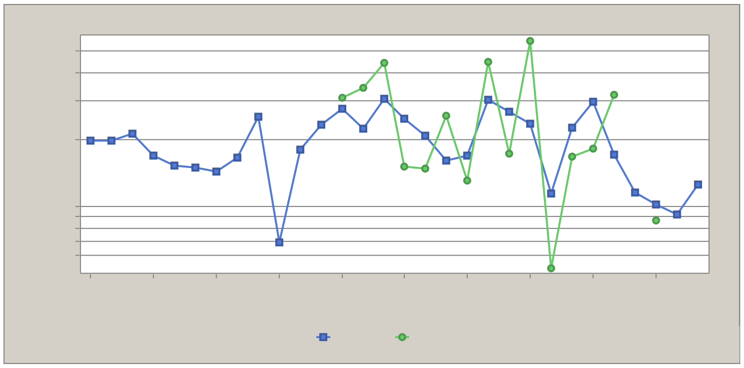
Recruitment Index 3, Linear and Log Scale



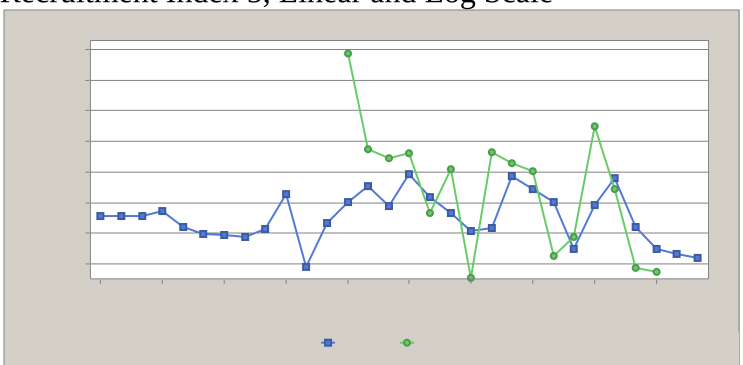


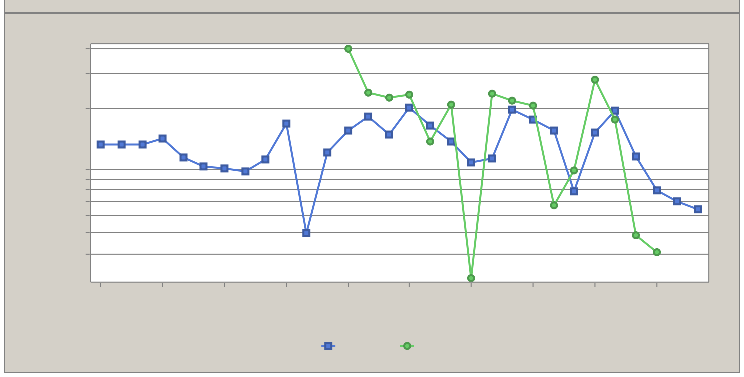
Recruitment Index 4, Linear and Log Scale



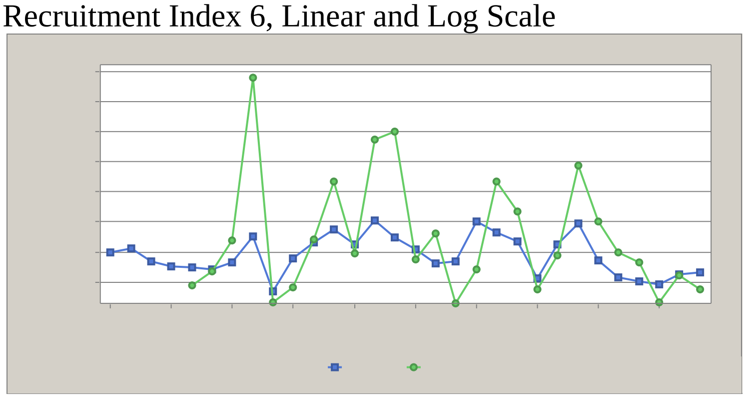


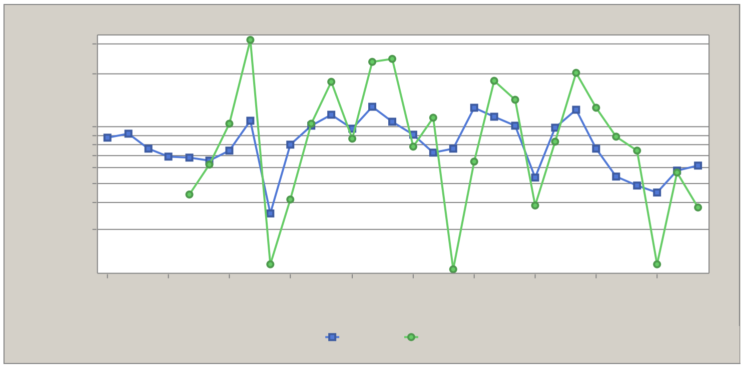
Recruitment Index 5, Linear and Log Scale



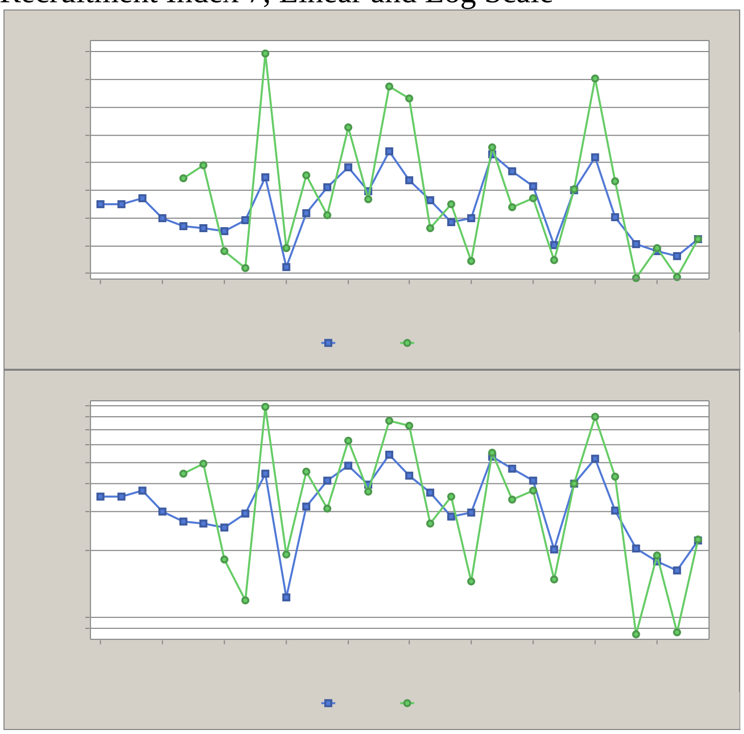


Recruitment Index 6, Linear and Log Scale

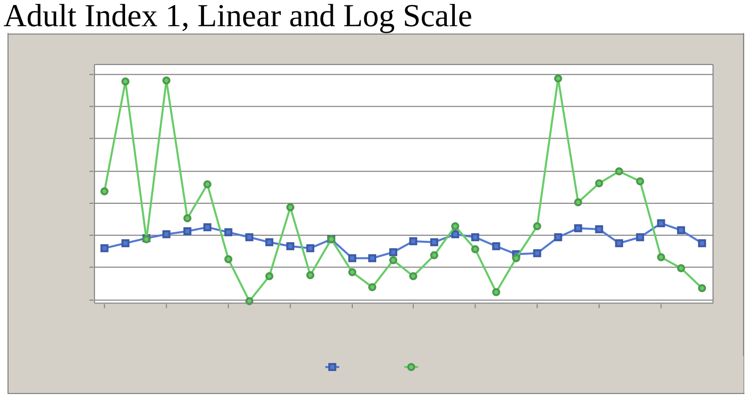


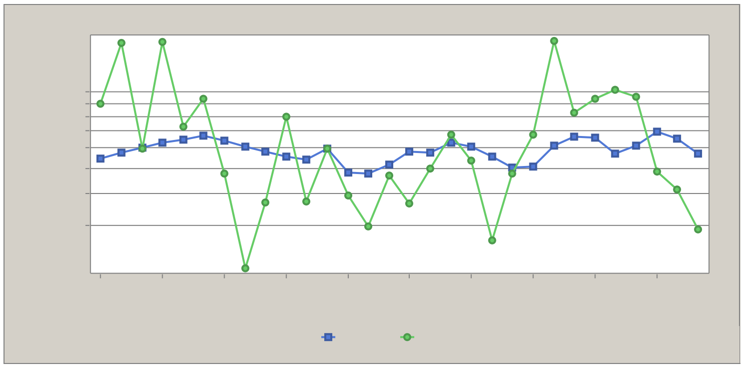


Recruitment Index 7, Linear and Log Scale



Adult Index 1, Linear and Log Scale





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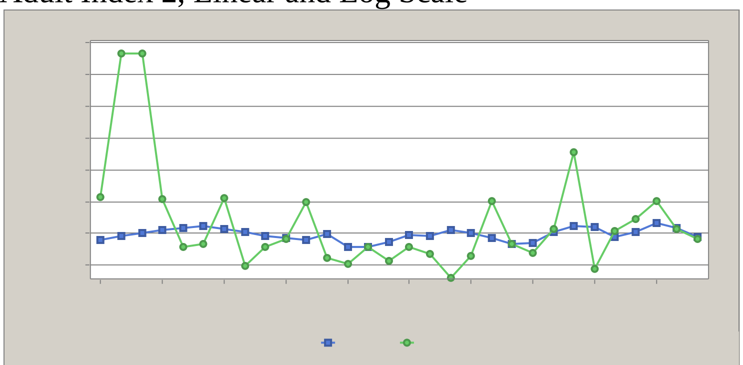
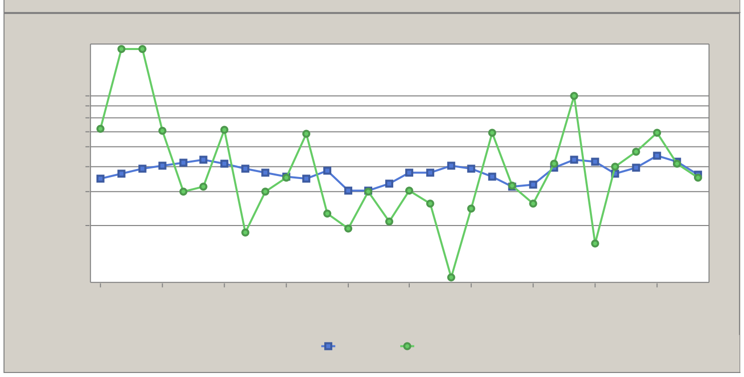
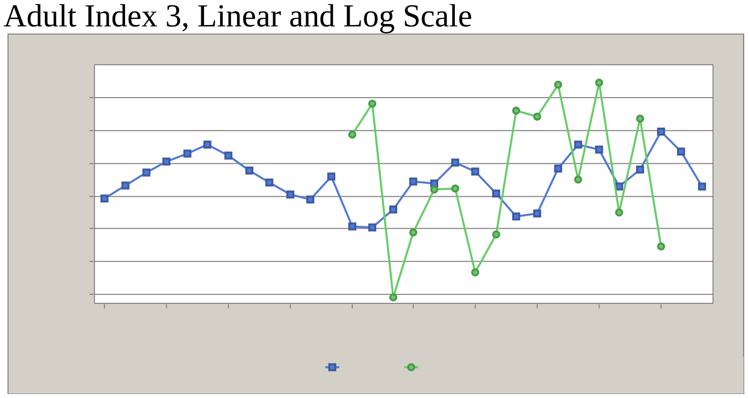
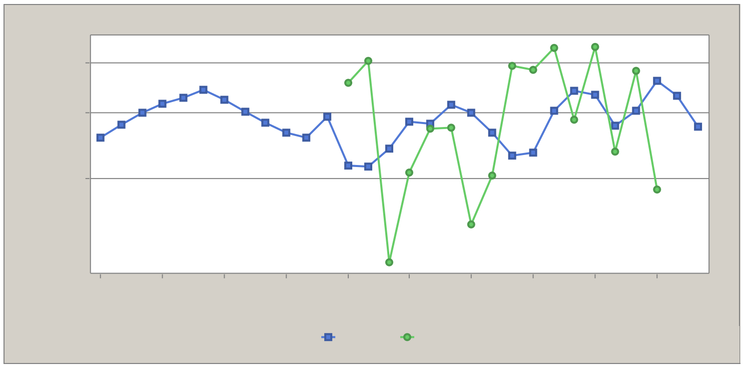


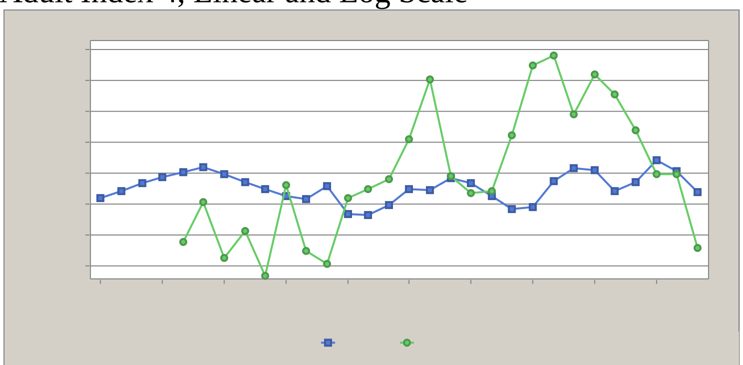


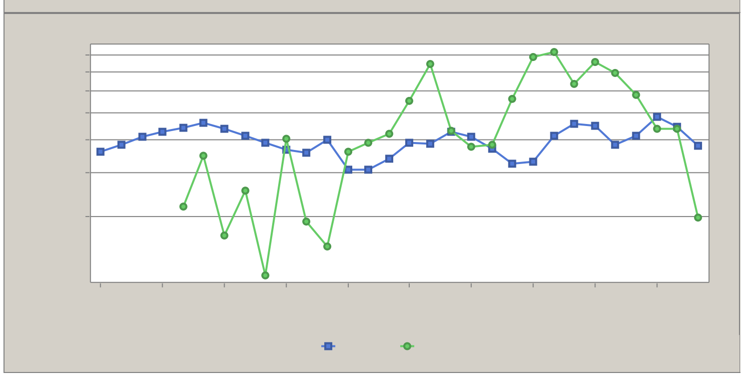
Adult Index 3, Linear and Log Scale



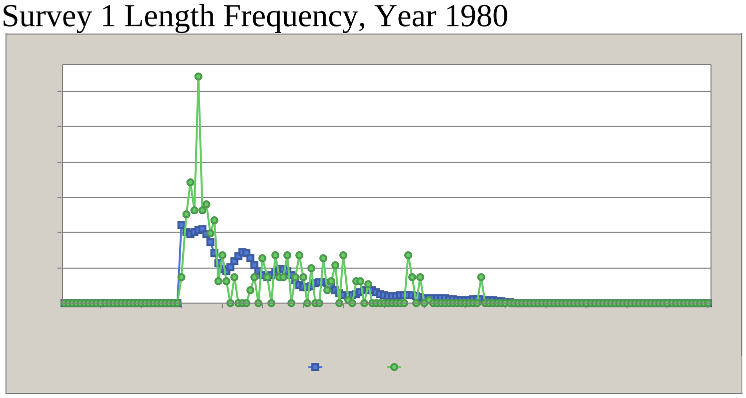


Adult Index 4, Linear and Log Scale





Survey 1 Length Frequency, Year 1980



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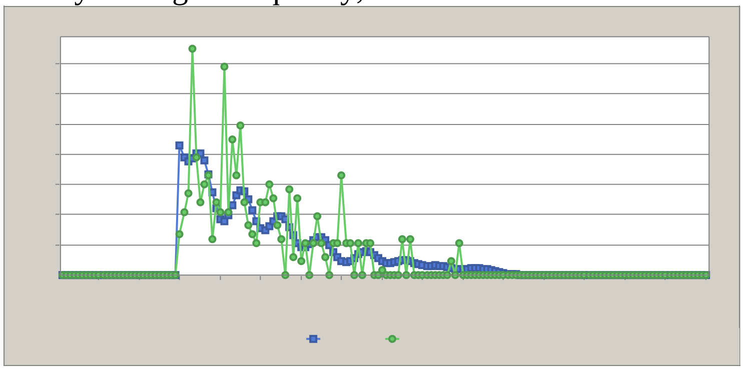

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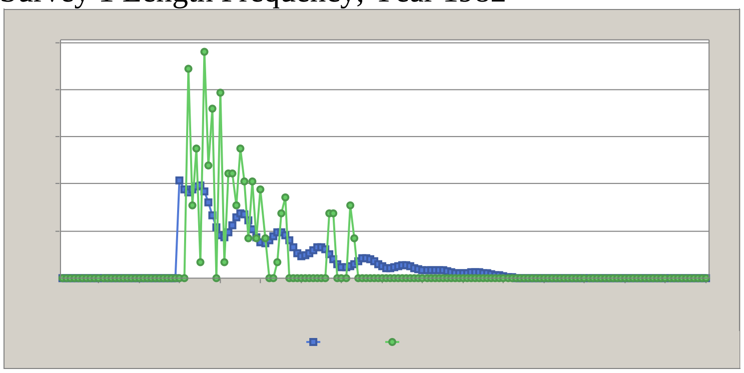

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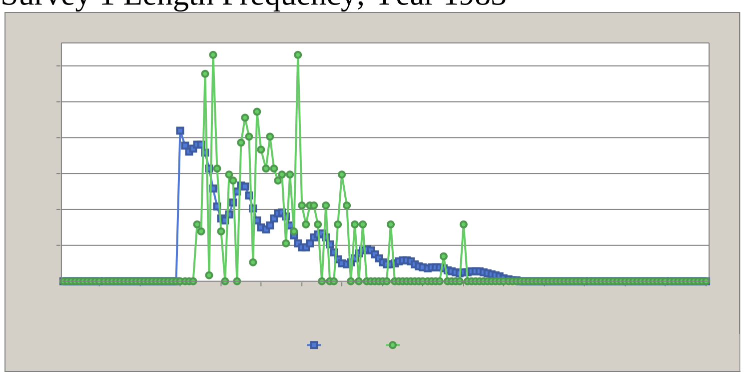

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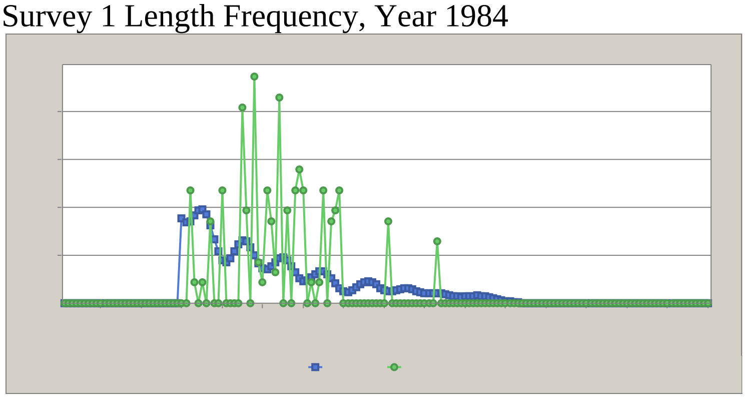

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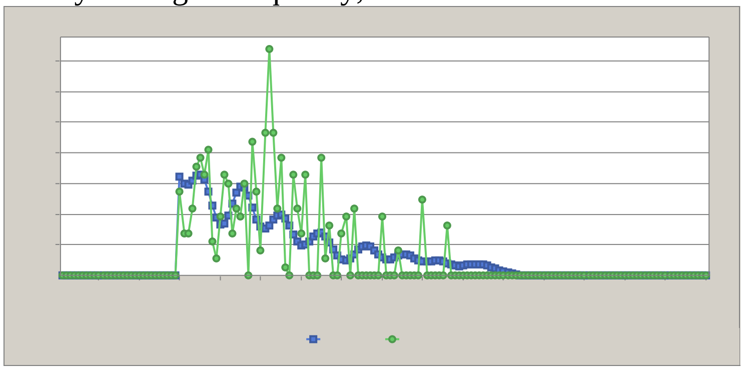

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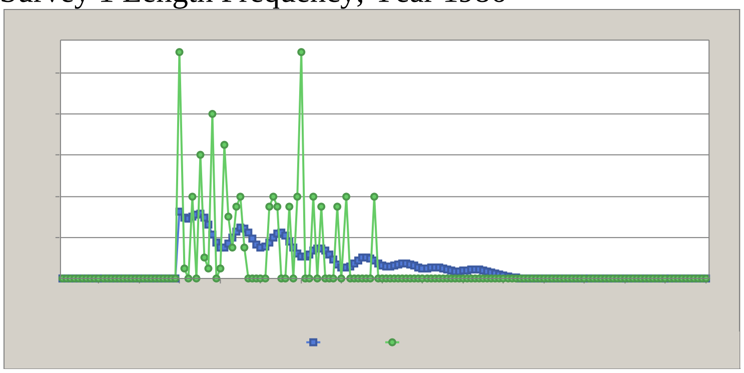

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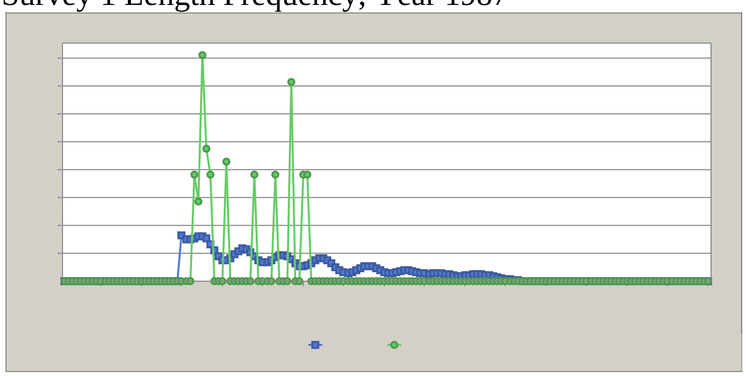

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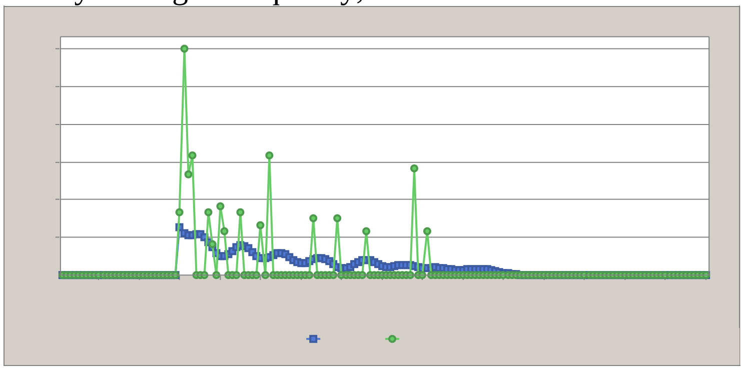

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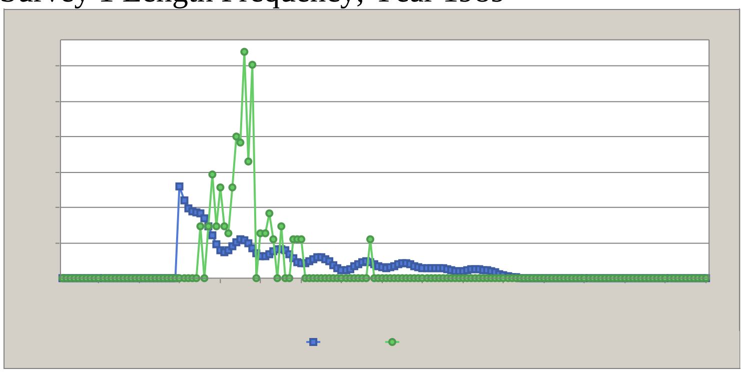

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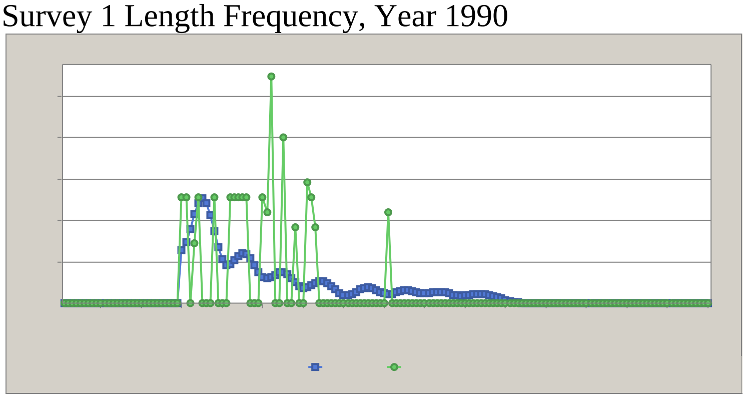

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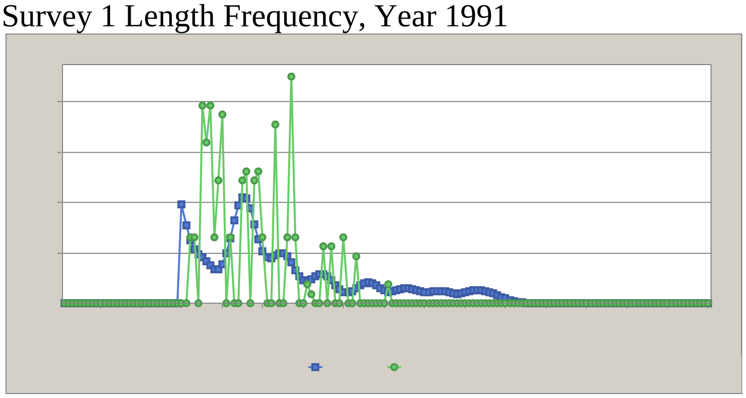

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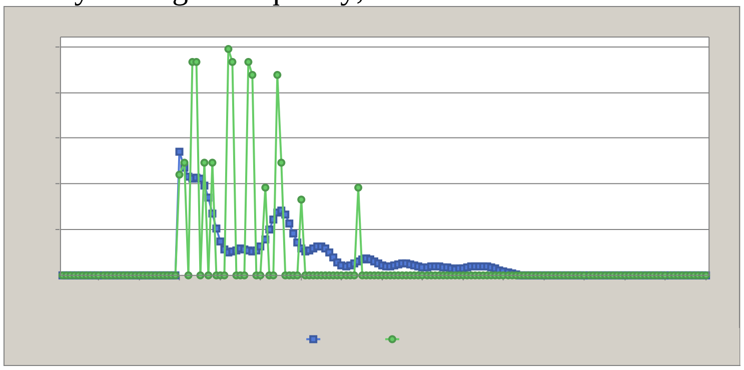

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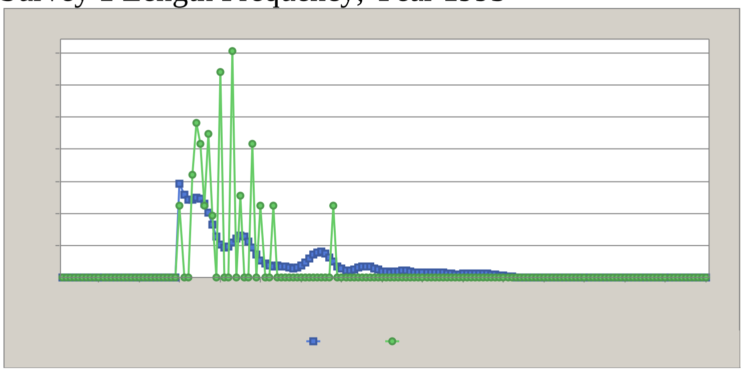

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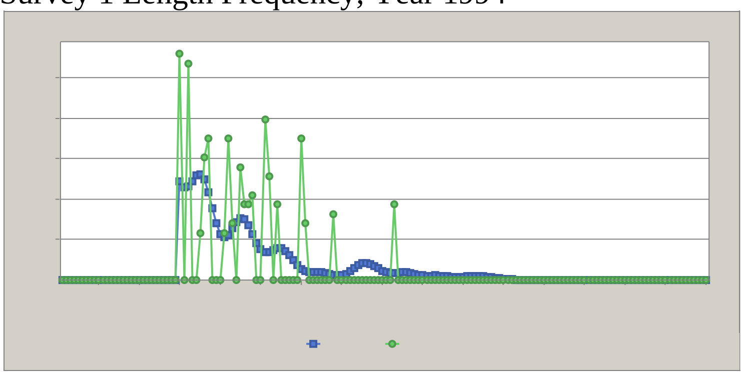

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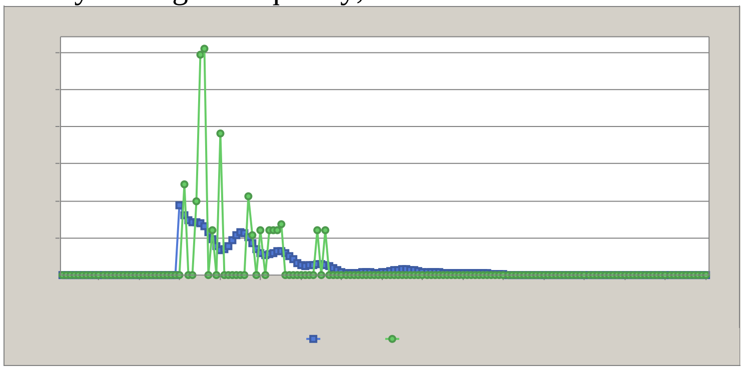

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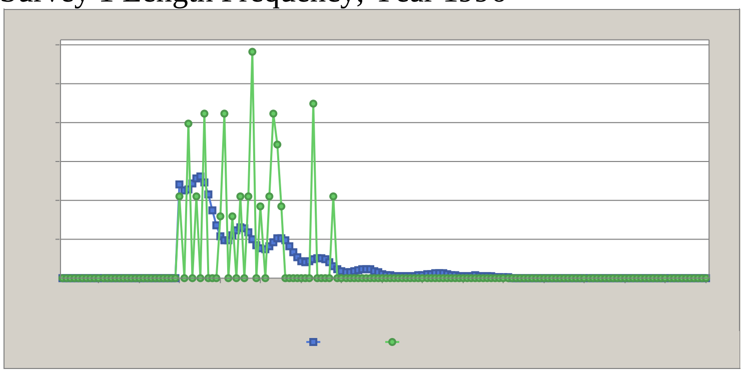

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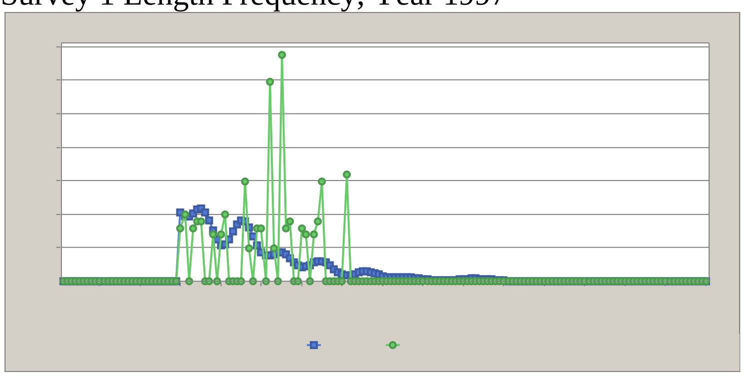

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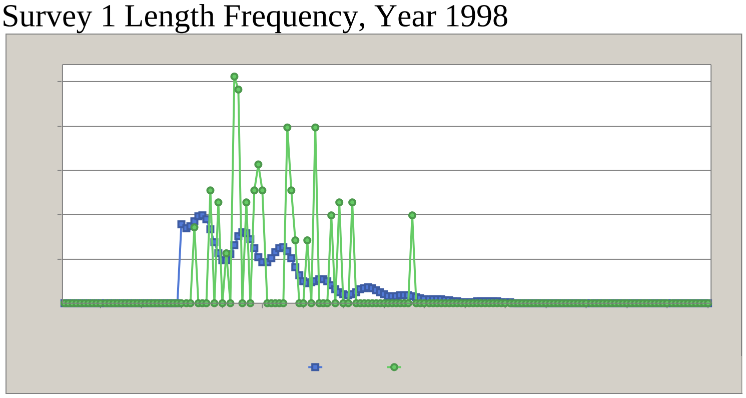

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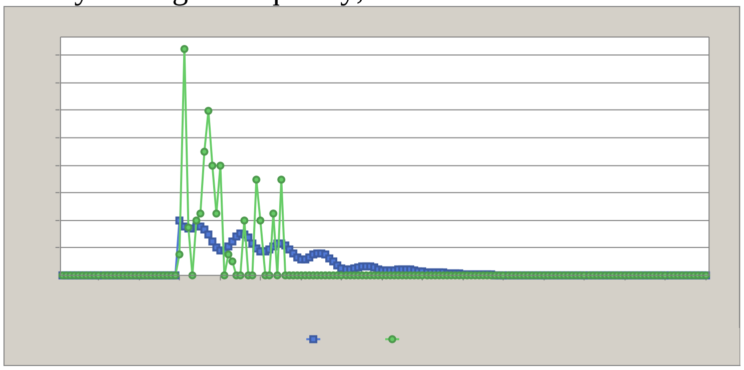
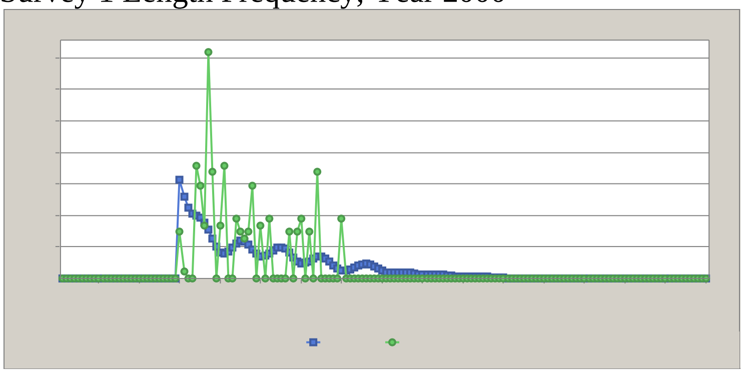


Survey 1 Length Frequency, Year 2000



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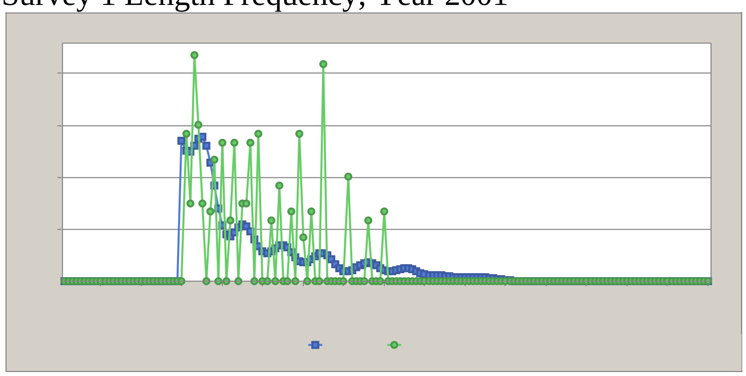
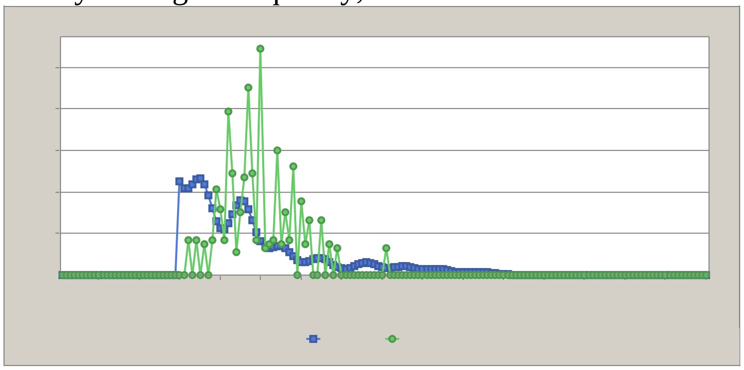


Survey 1 Length Frequency, Year 2002



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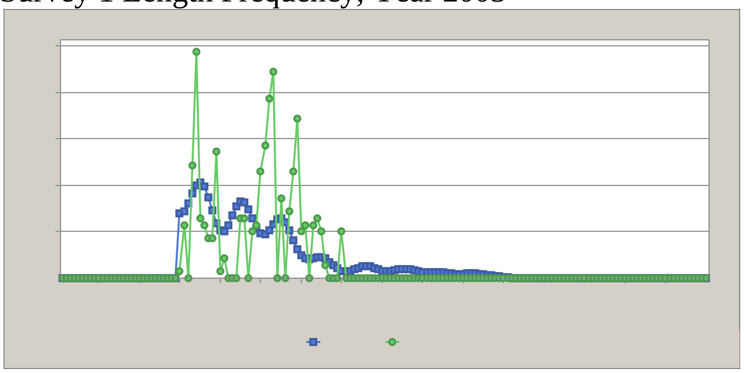

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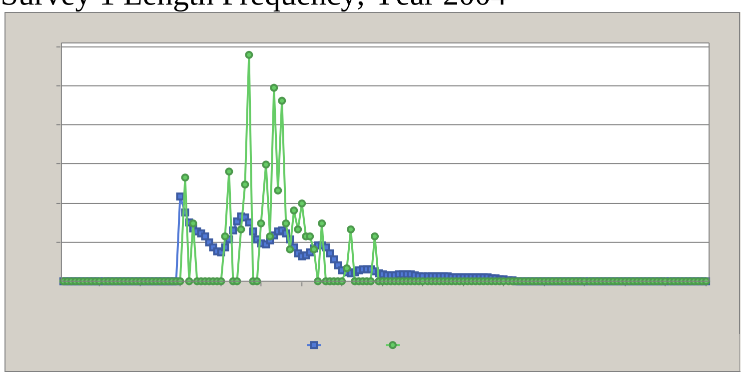

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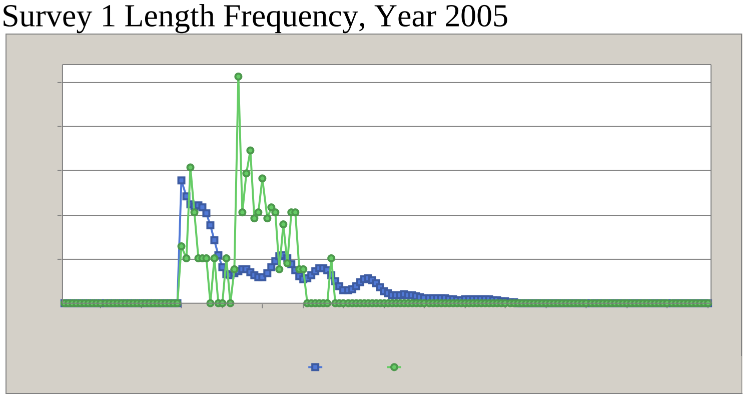

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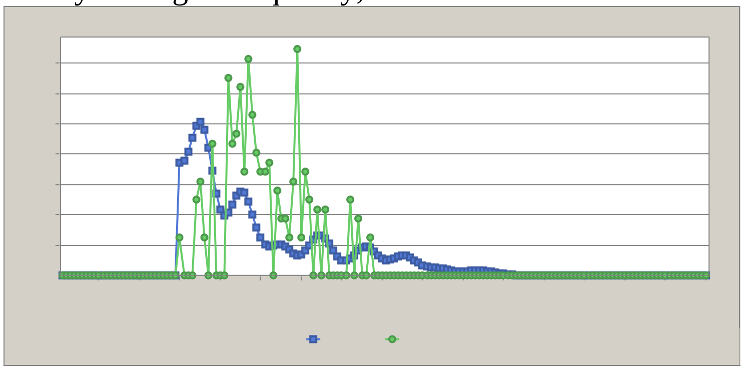
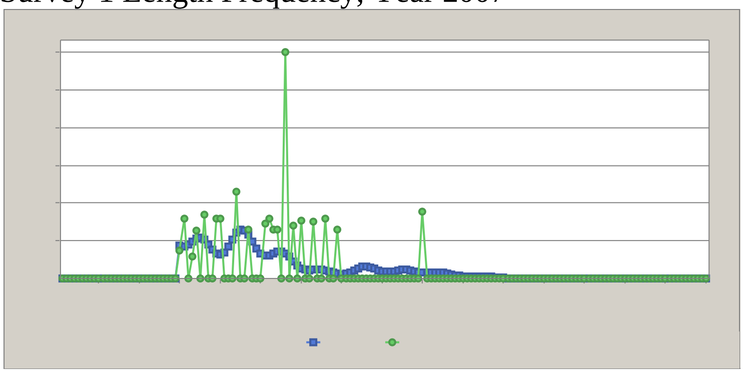
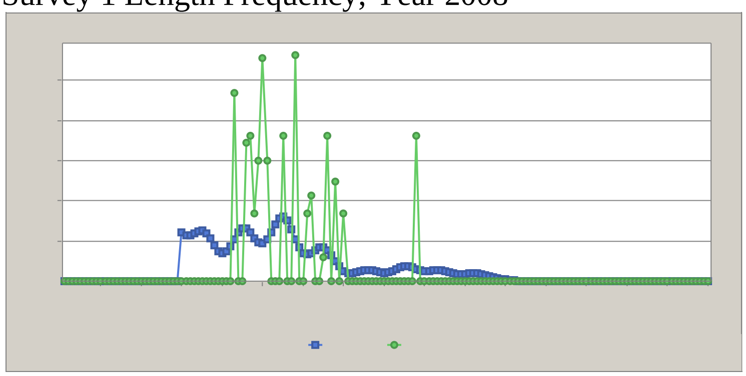


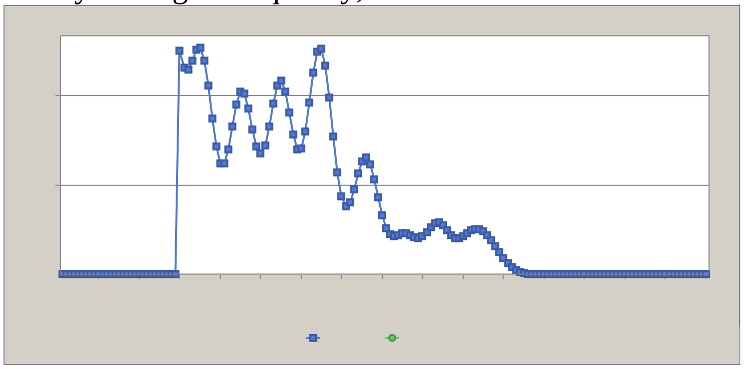
Survey 1 Length Frequency, Year 2007



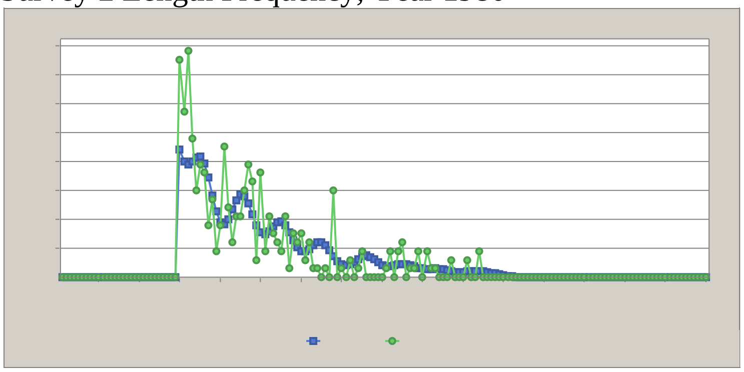
Survey 1 Length Frequency, Year 2008



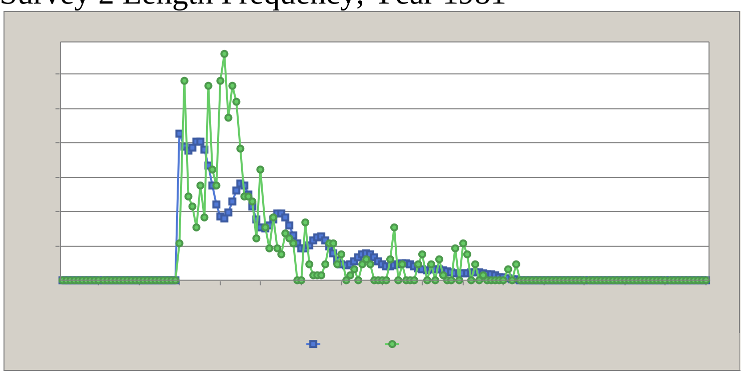
Survey 1 Length Frequency, Year 2009



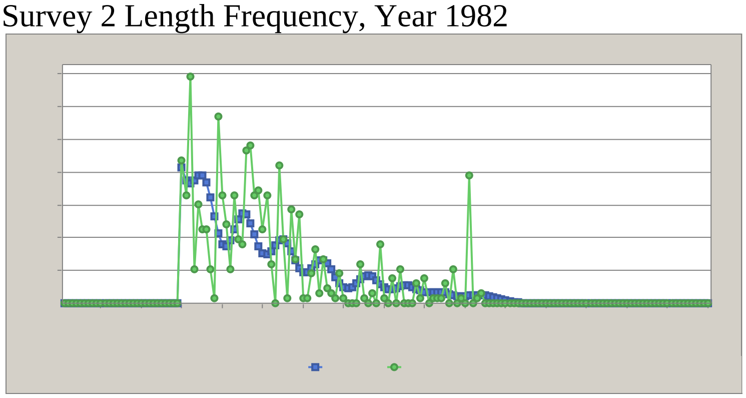
Survey 2 Length Frequency, Year 1980



Survey 2 Length Frequency, Year 1981



Survey 2 Length Frequency, Year 1982



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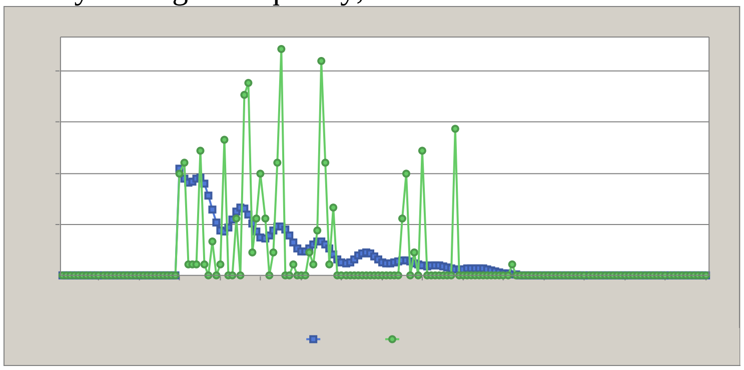
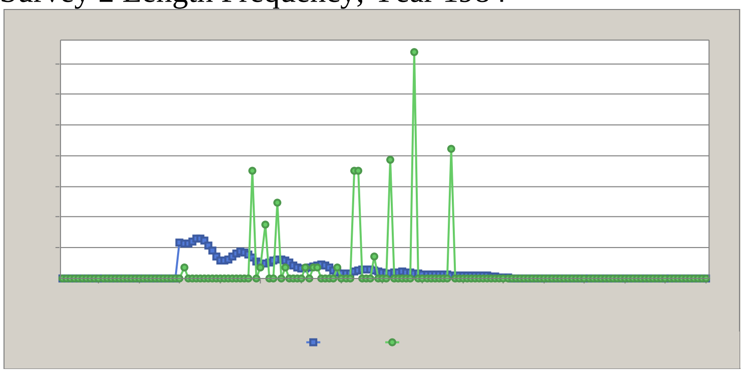


Survey 2 Length Frequency, Year 1984



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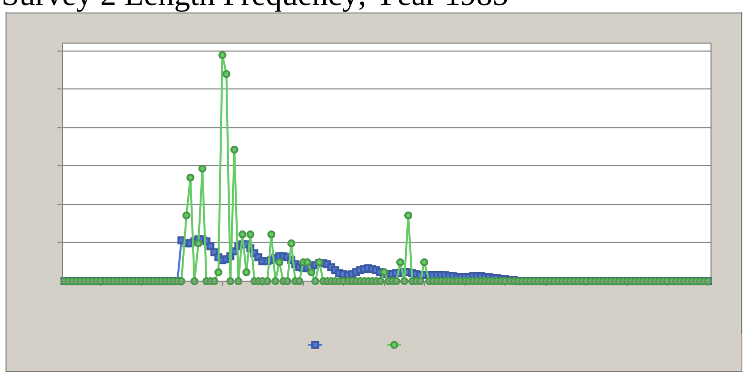

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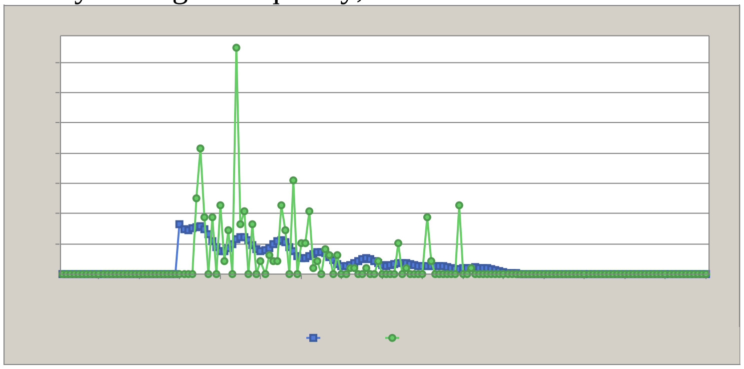

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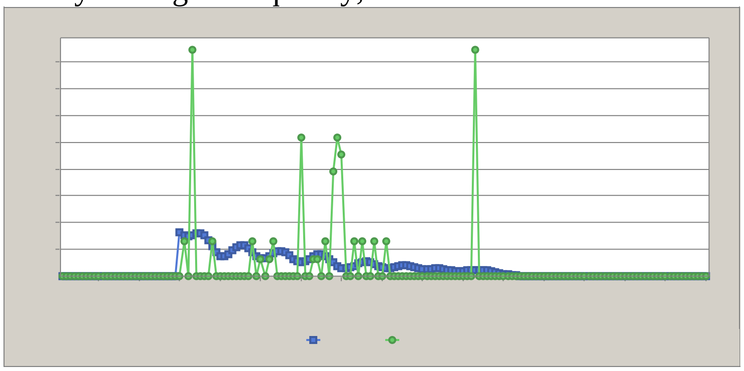

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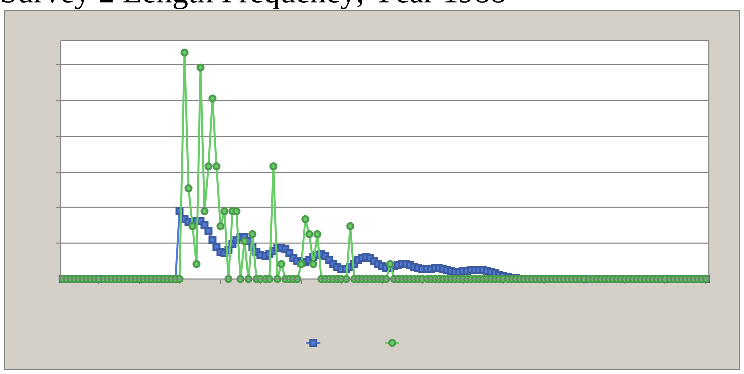
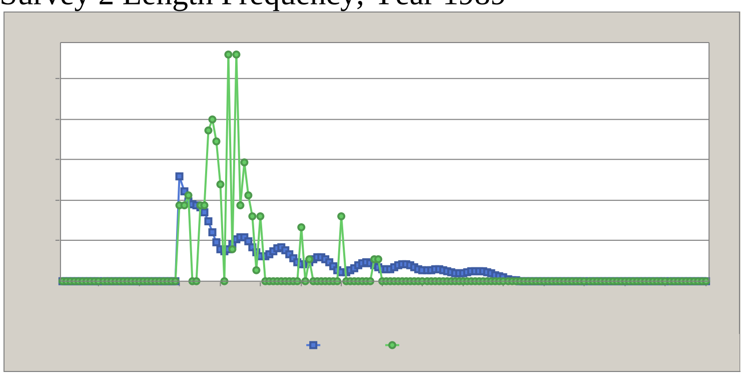


Survey 2 Length Frequency, Year 1989



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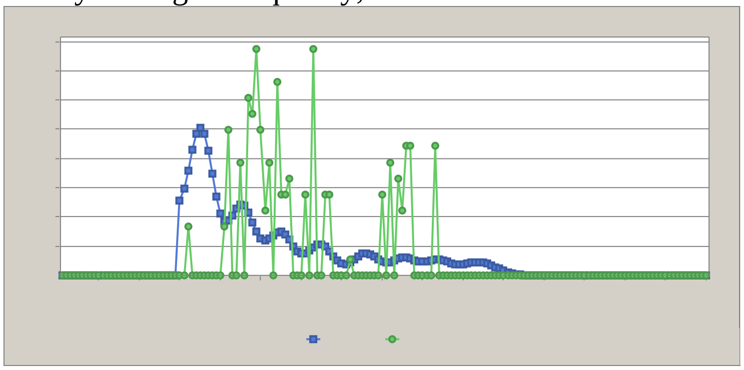

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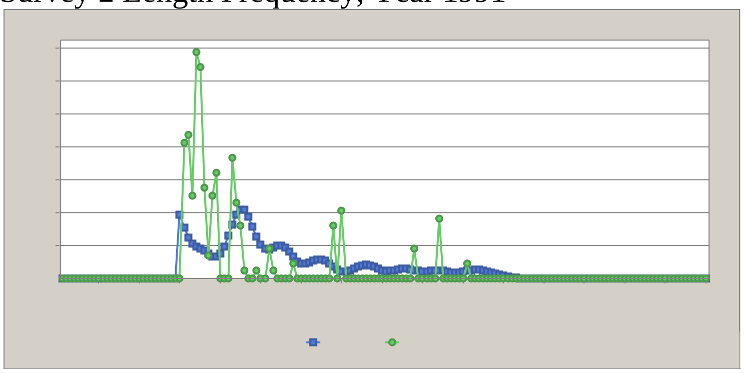

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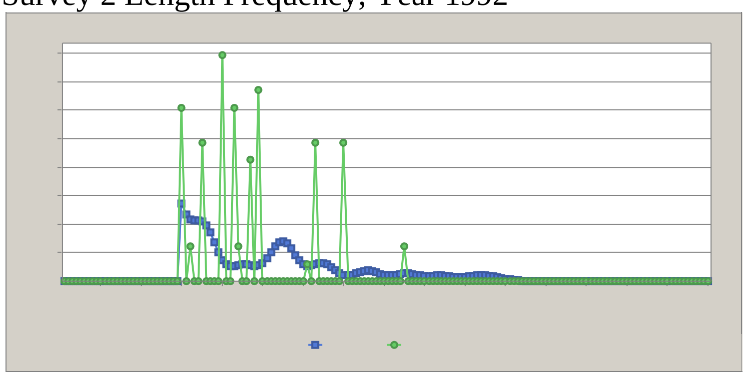

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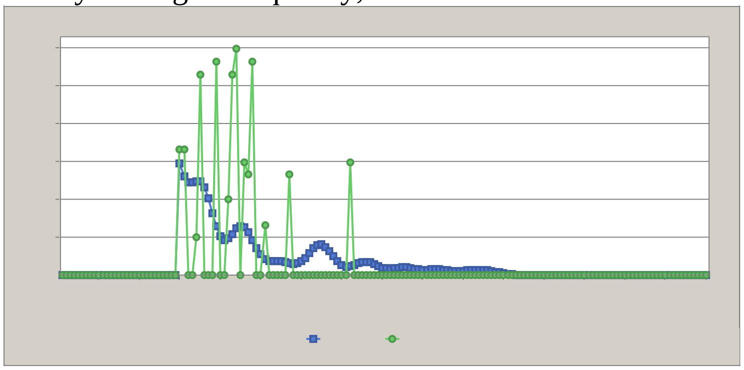

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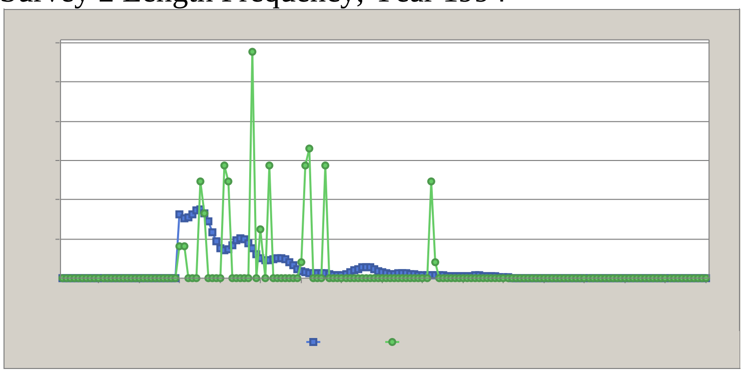

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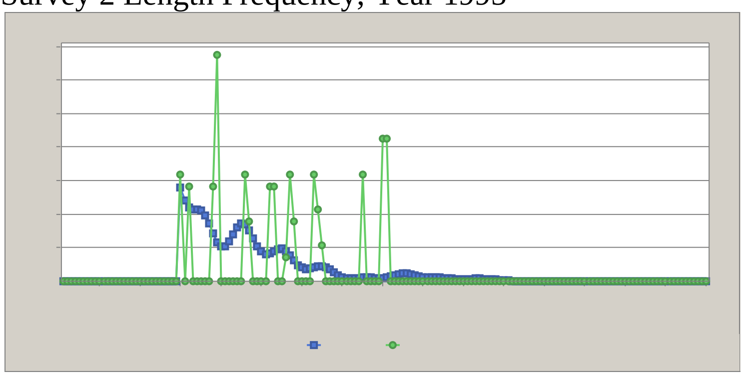
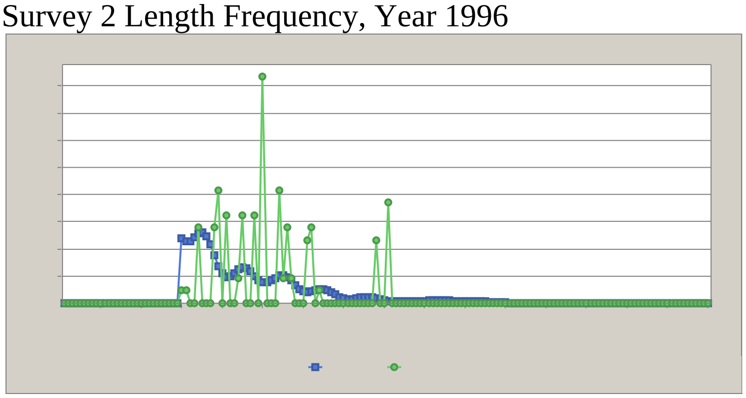


Survey 2 Length Frequency, Year 1996



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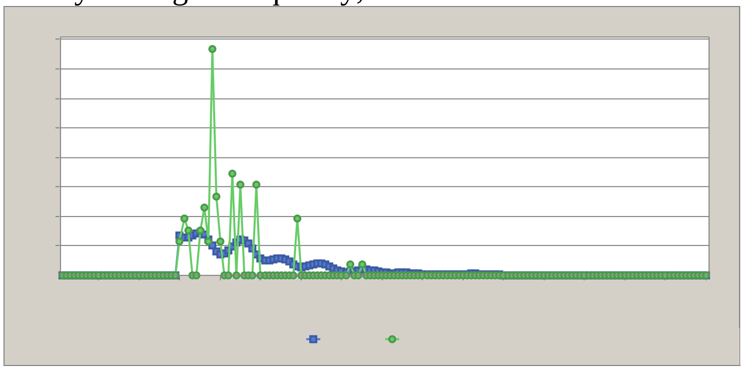

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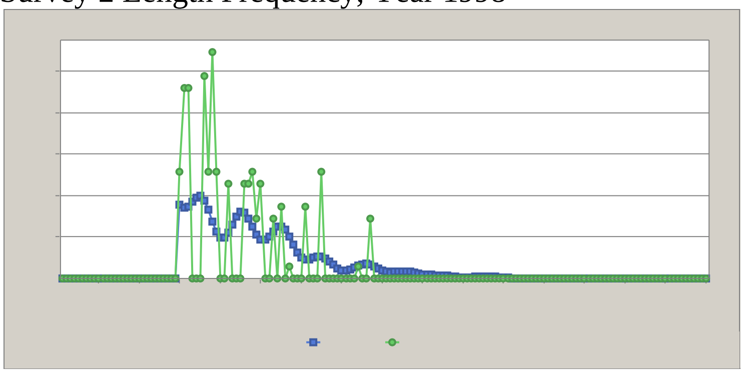

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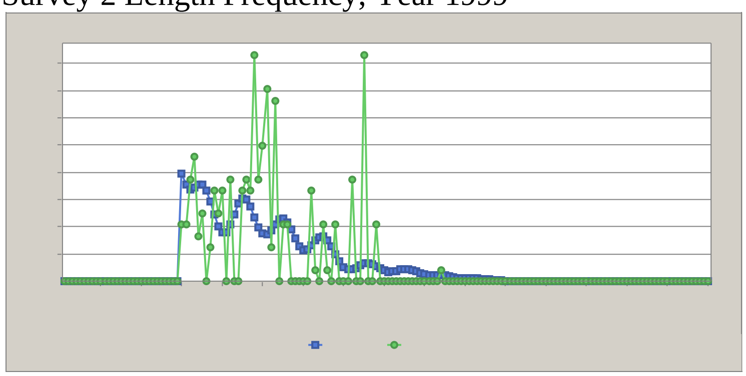

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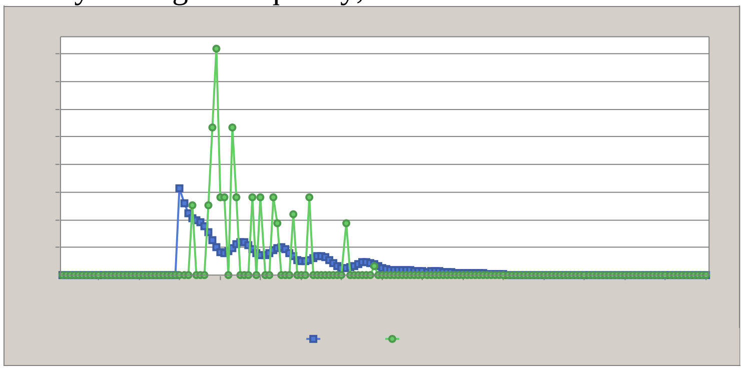

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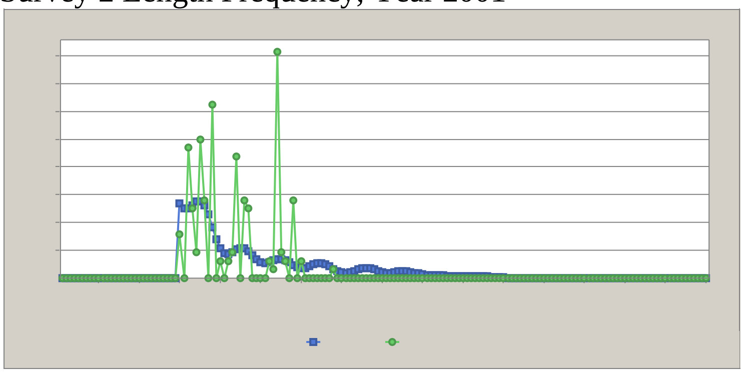

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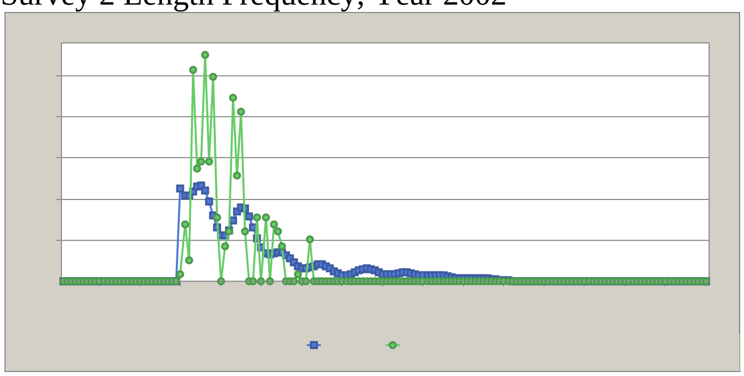

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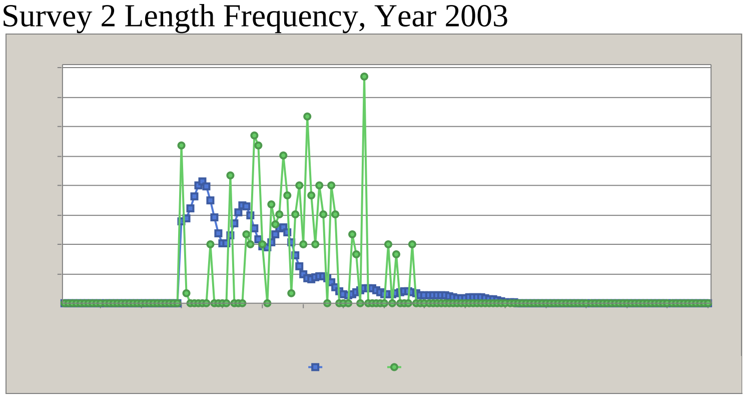

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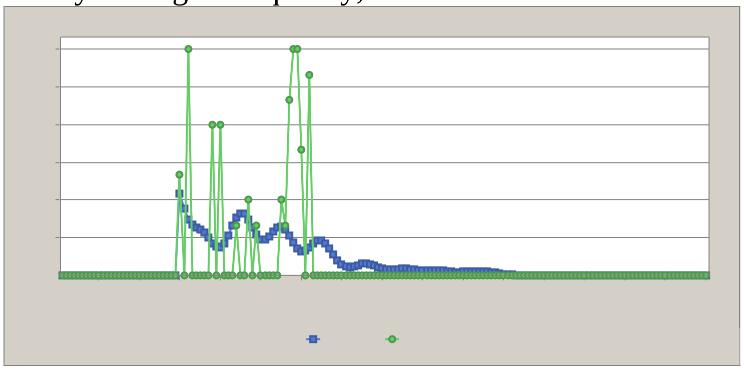

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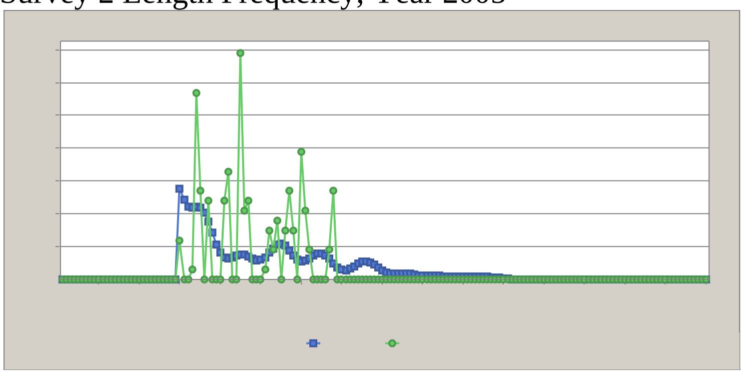

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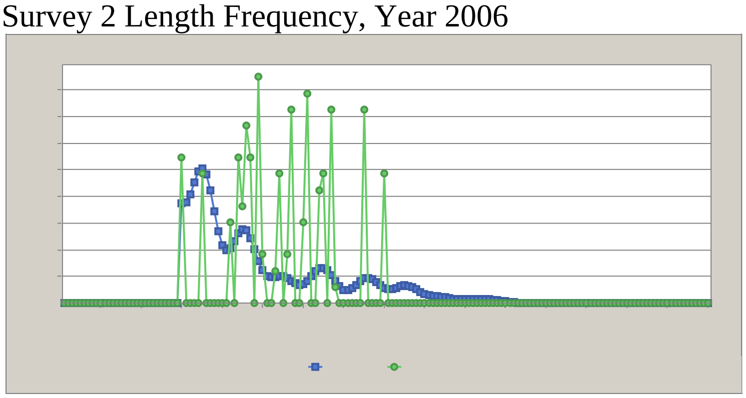

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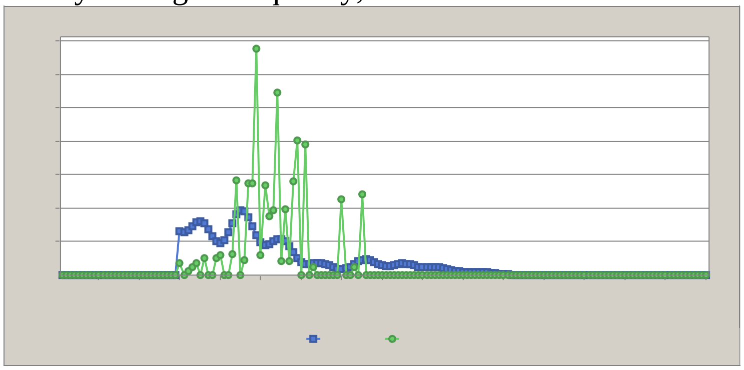
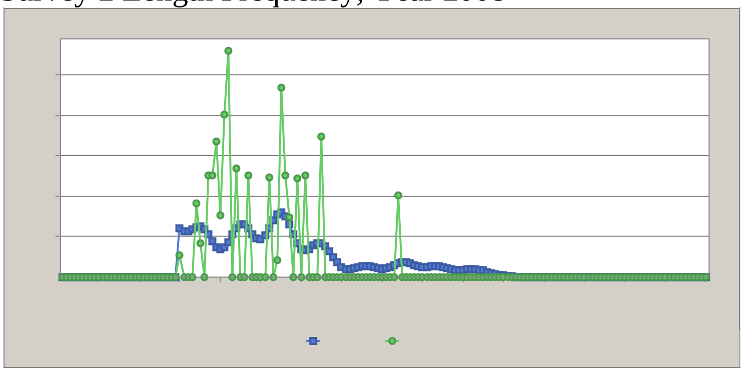
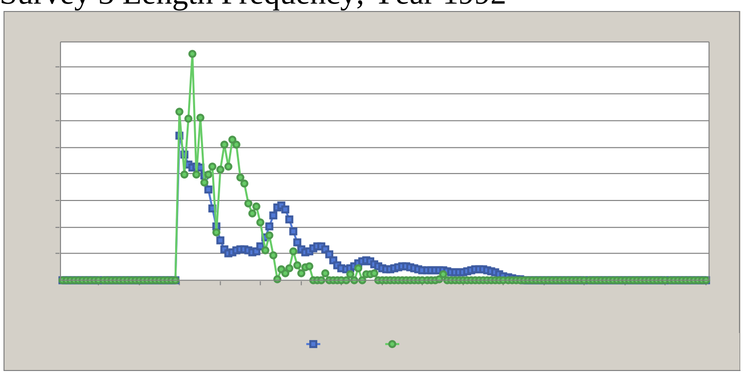


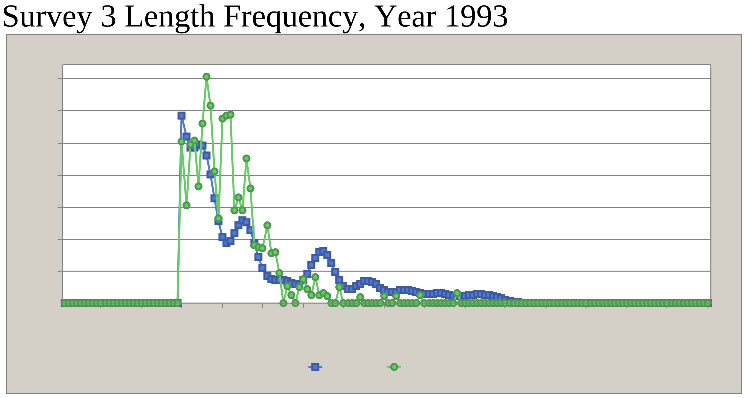
Survey 2 Length Frequency, Year 2008



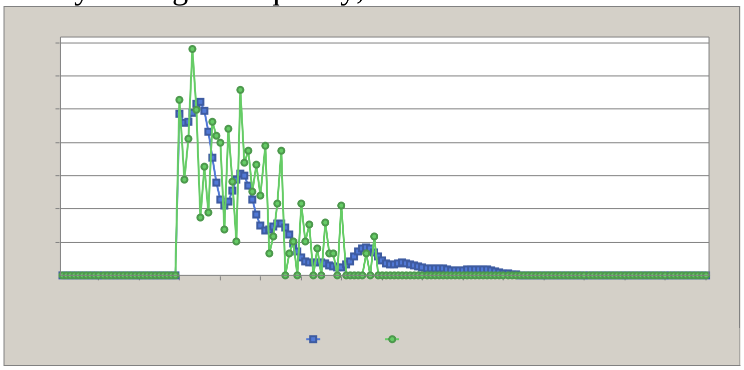
Survey 3 Length Frequency, Year 1992



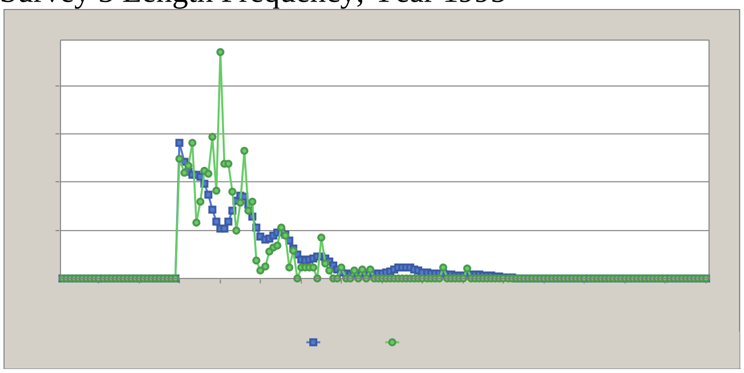
Survey 3 Length Frequency, Year 1993



Survey 3 Length Frequency, Year 1994



Survey 3 Length Frequency, Year 1995



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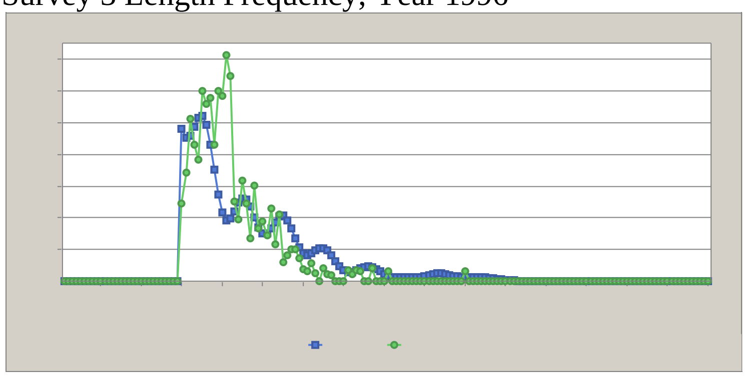

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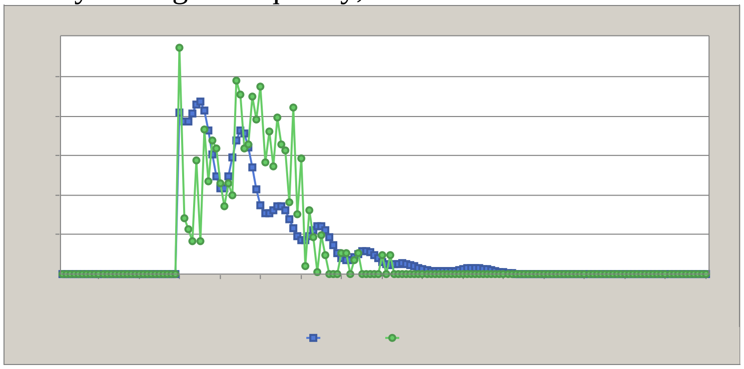

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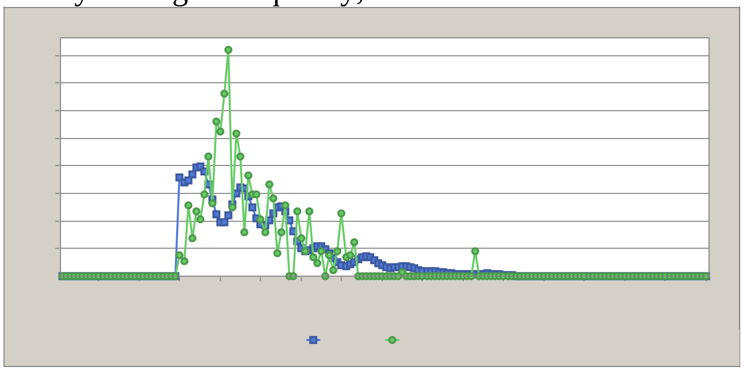

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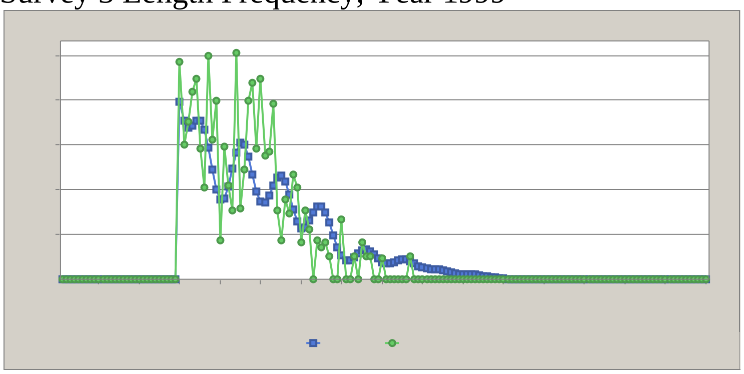

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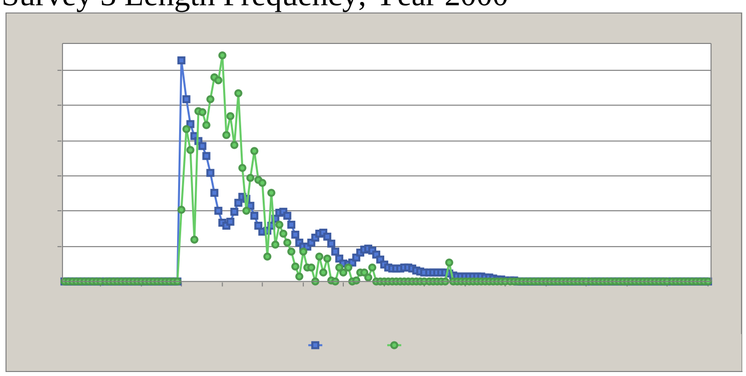

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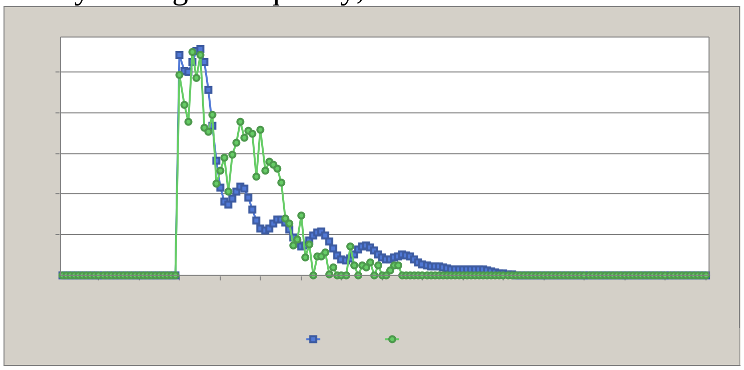

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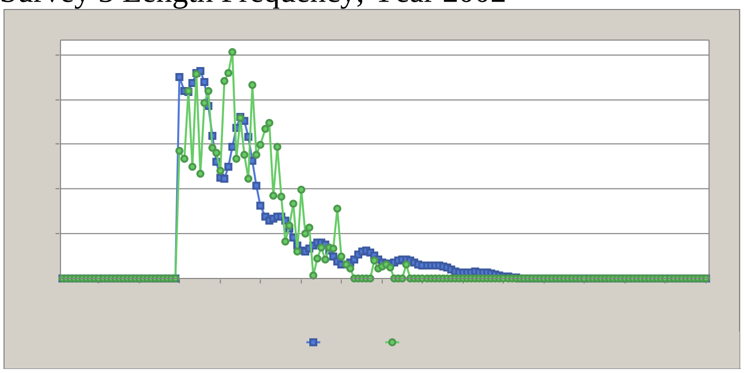

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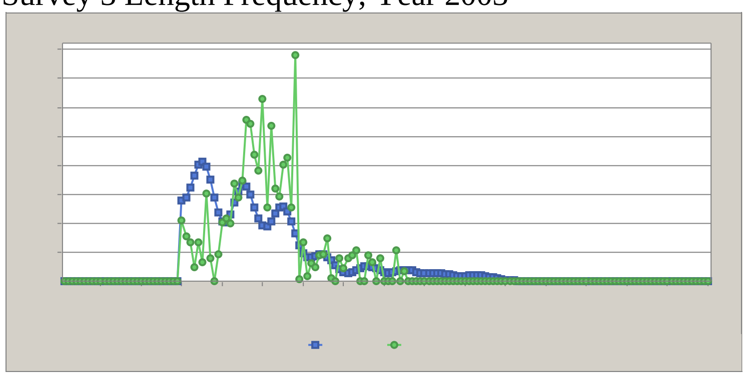

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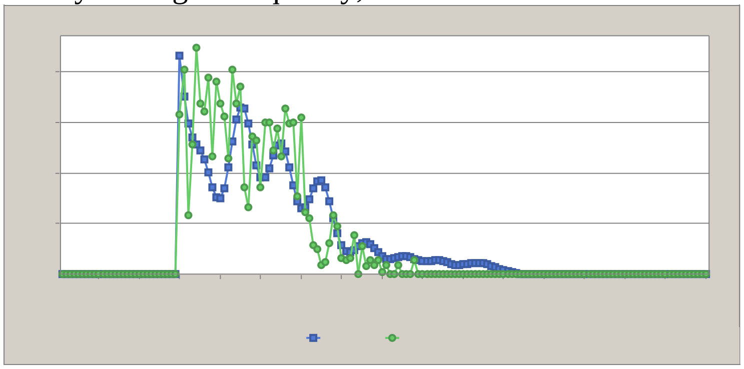

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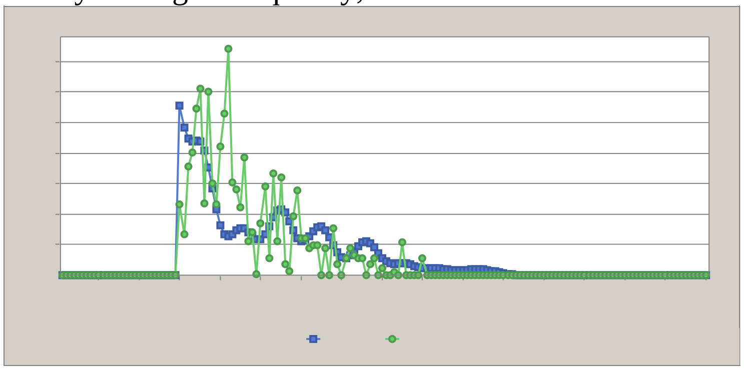

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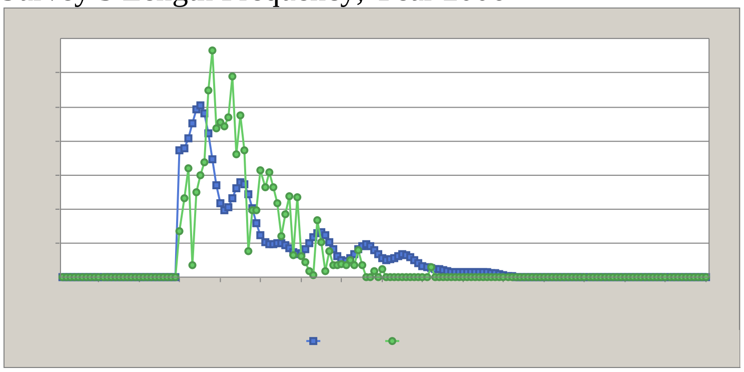

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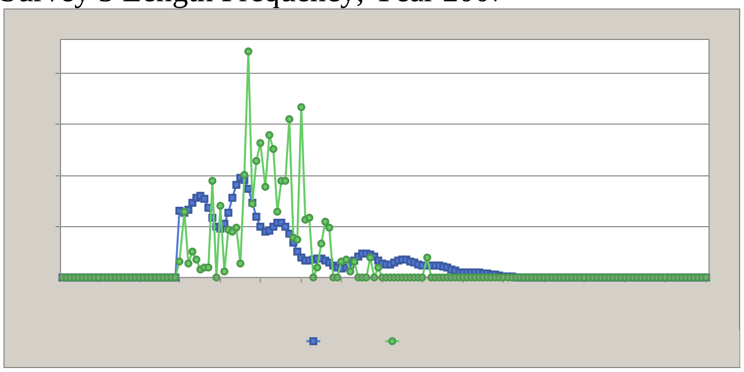
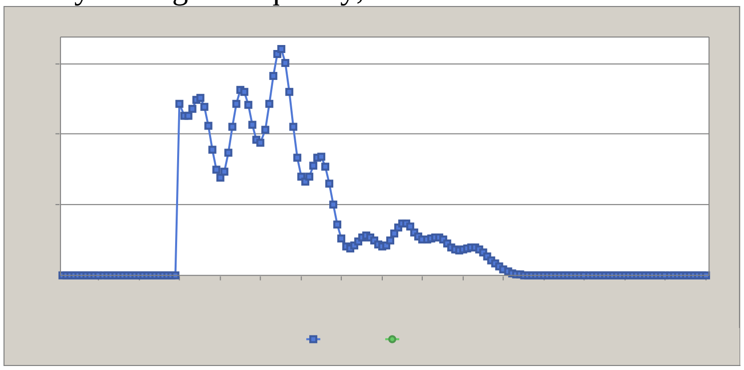


Survey 3 Length Frequency, Year 2008



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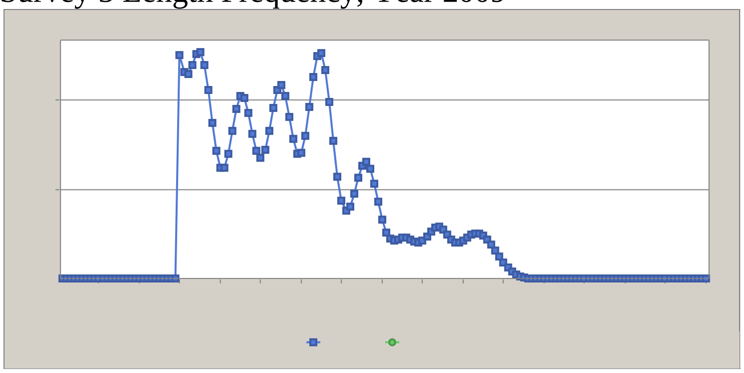

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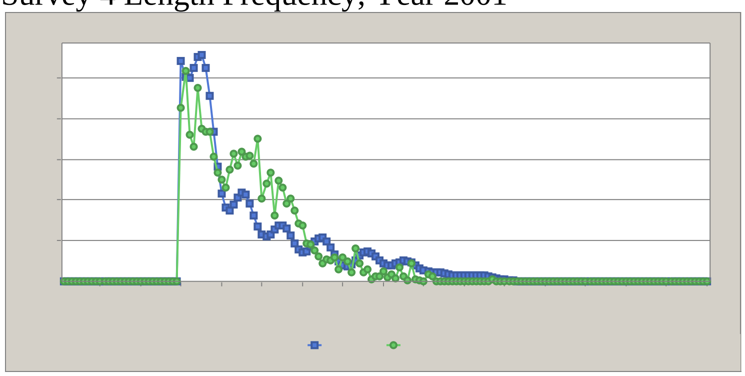
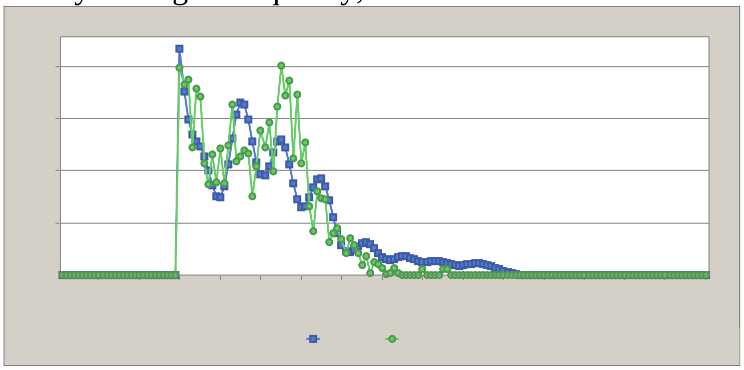


Survey 4 Length Frequency, Year 2004



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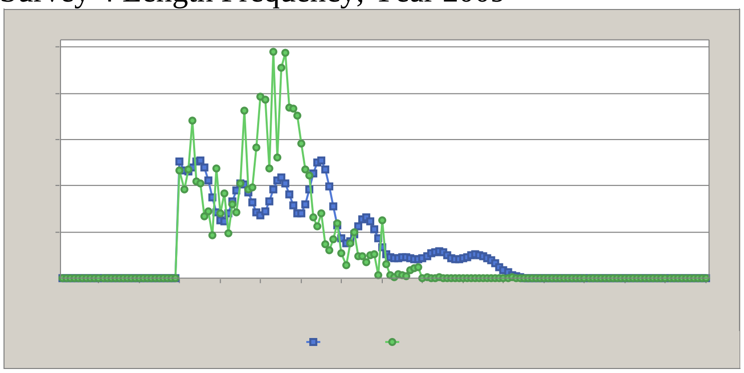
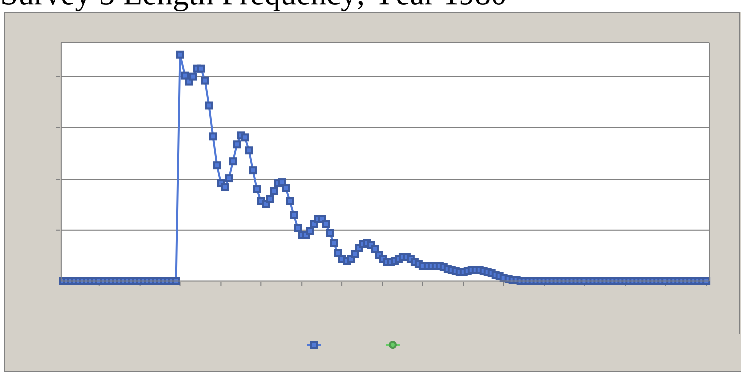
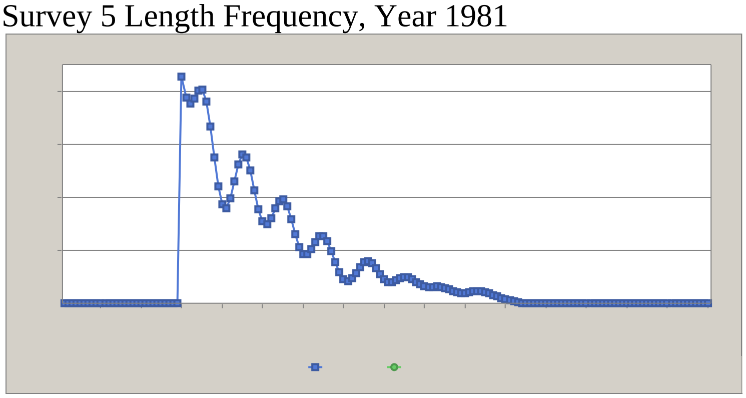


Survey 5 Length Frequency, Year 1980



Survey 5 Length Frequency, Year 1981



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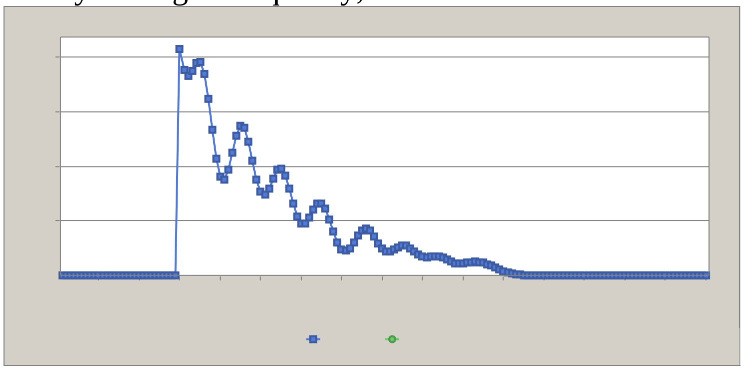

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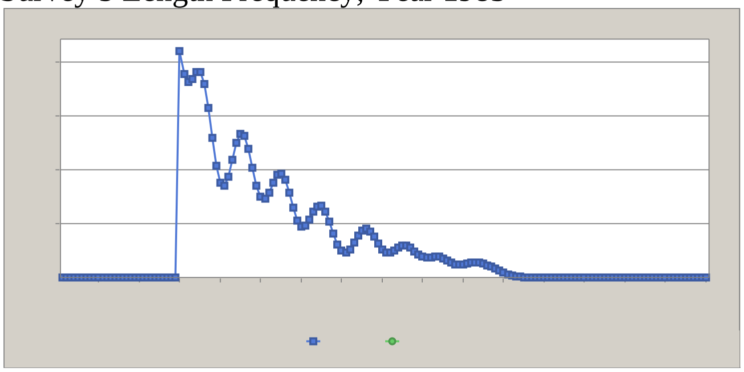

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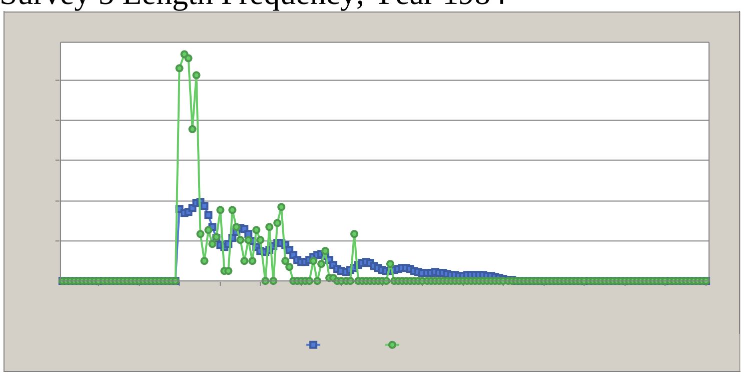

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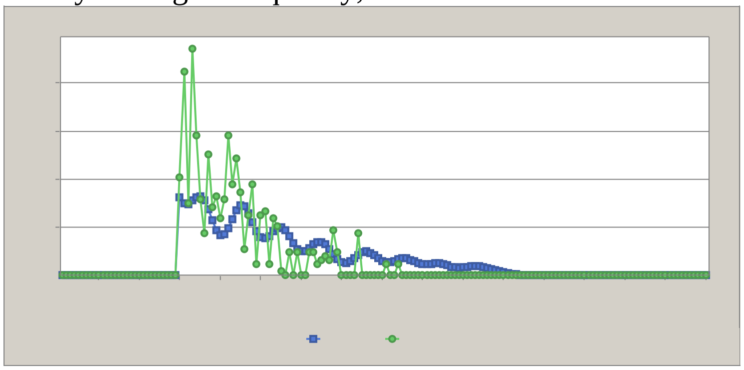

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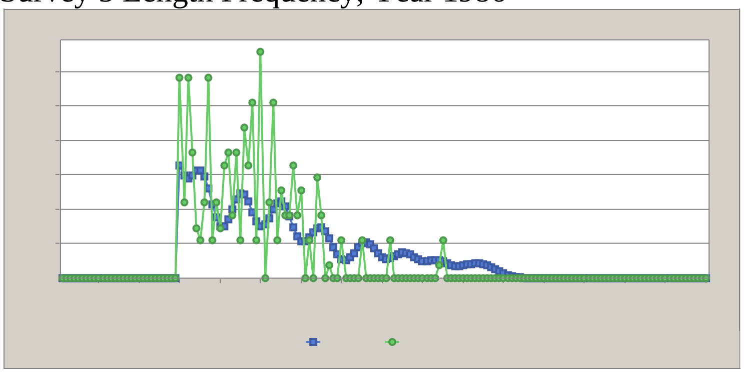

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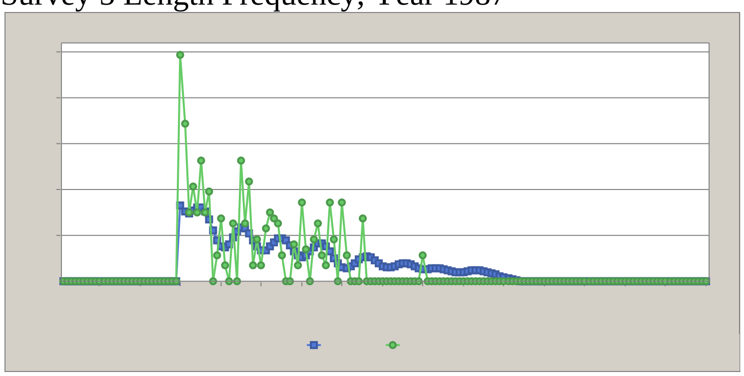

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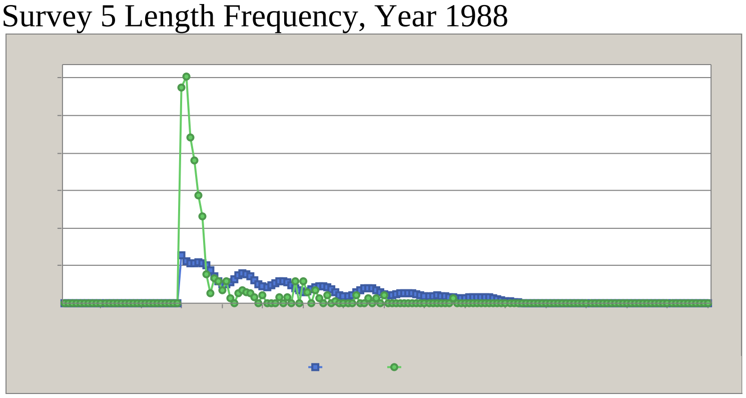

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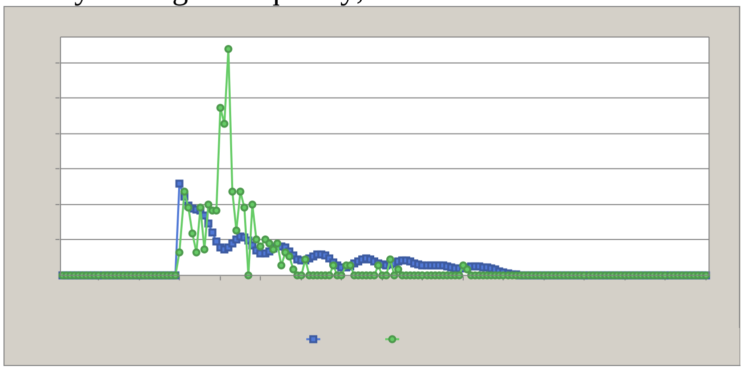

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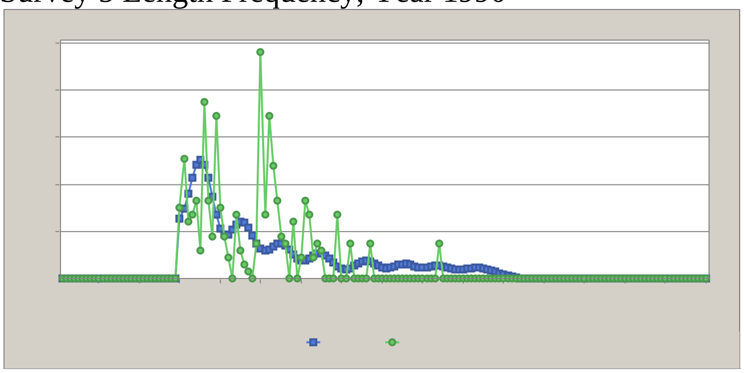

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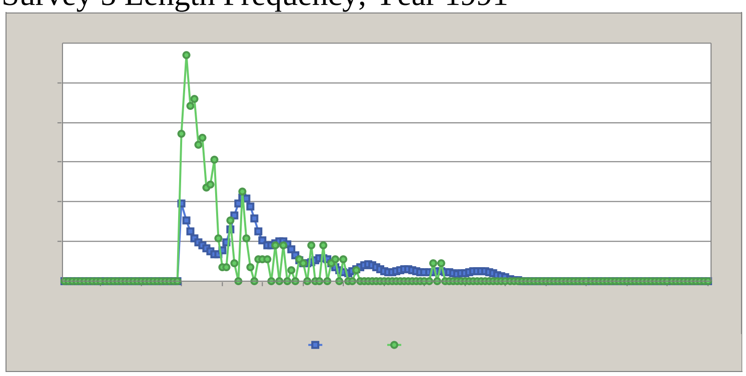

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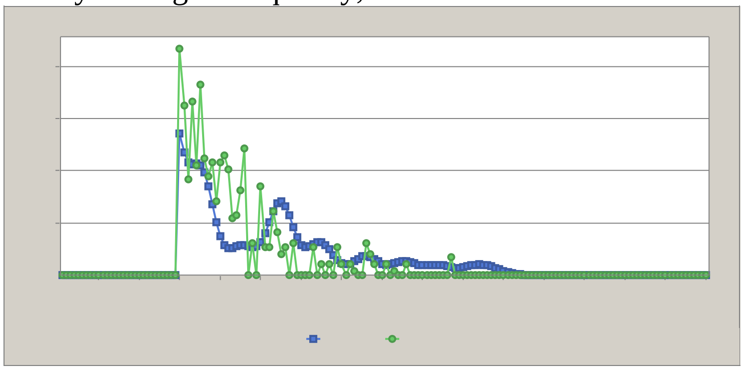

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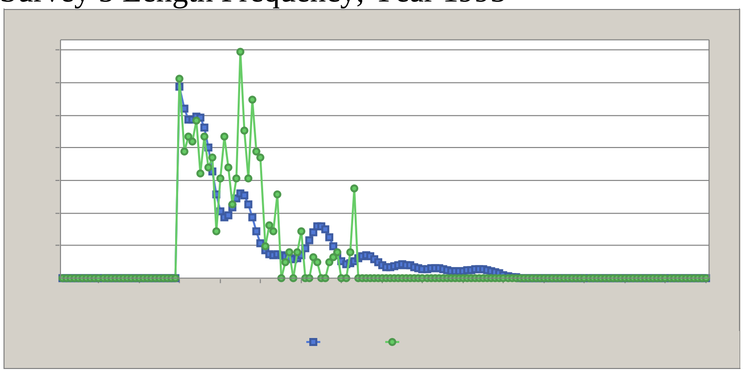

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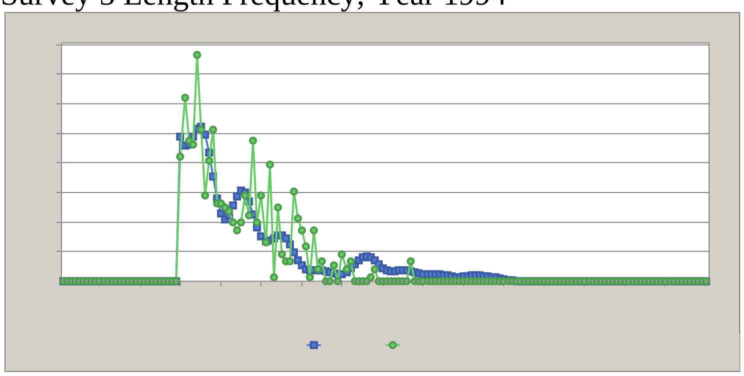

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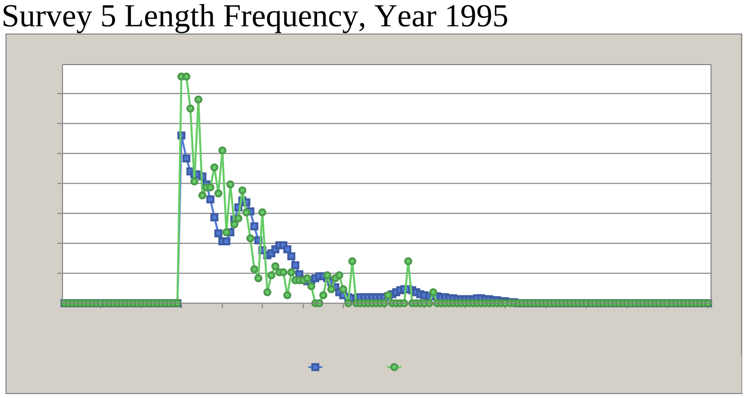

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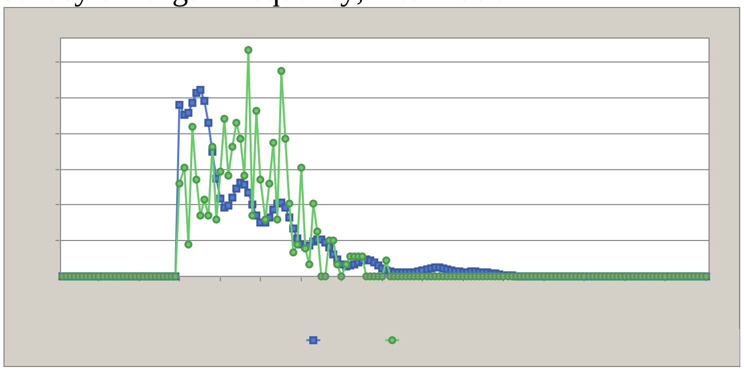

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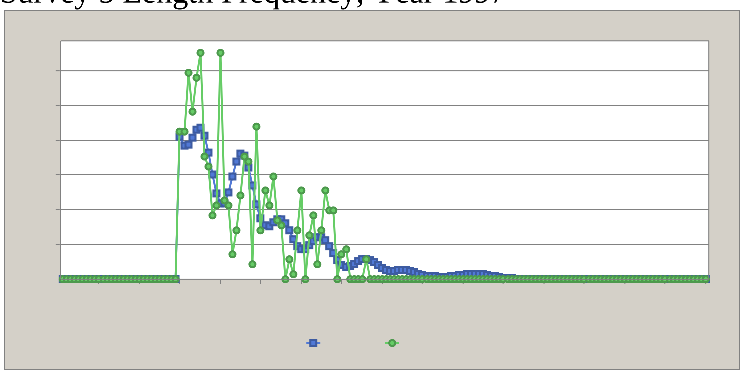

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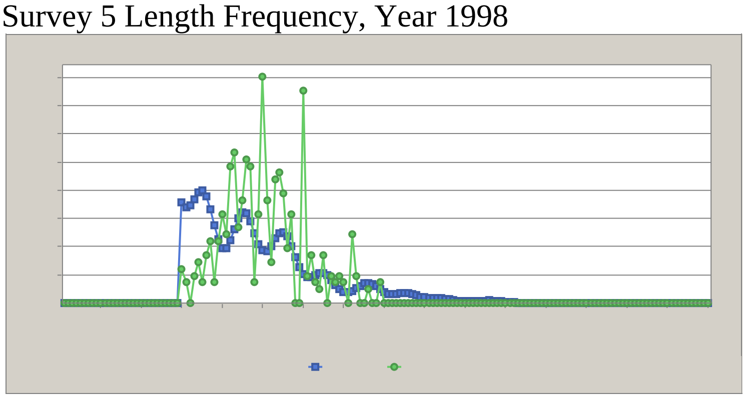

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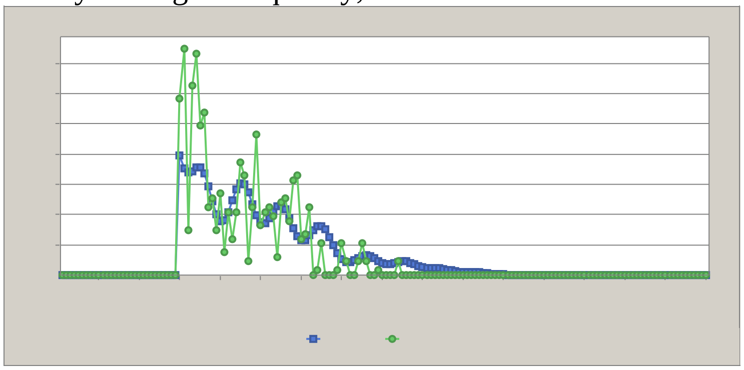
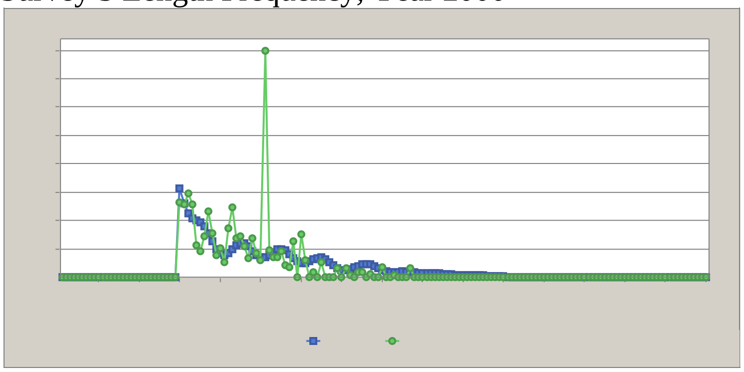
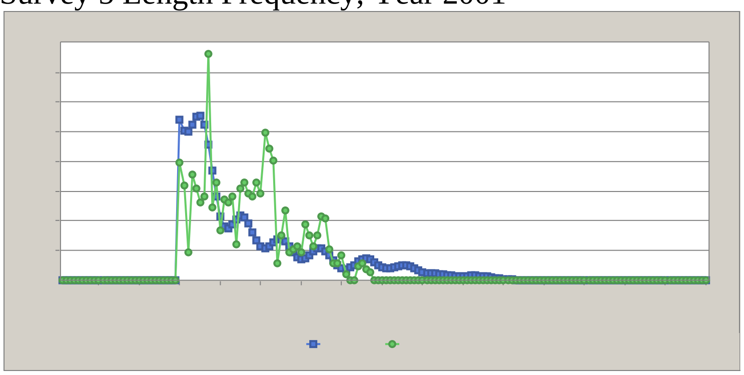


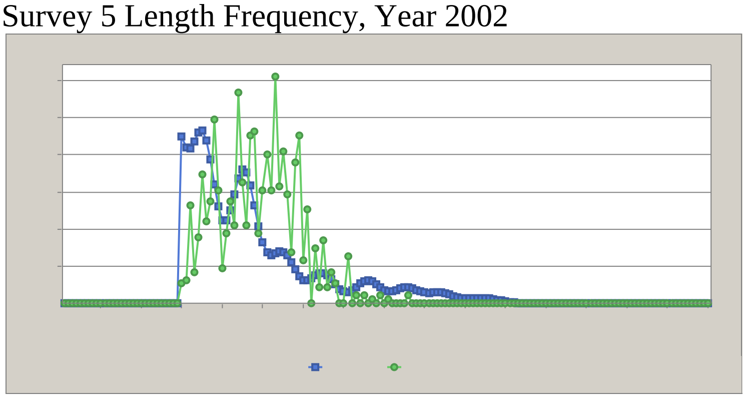
Survey 5 Length Frequency, Year 2000



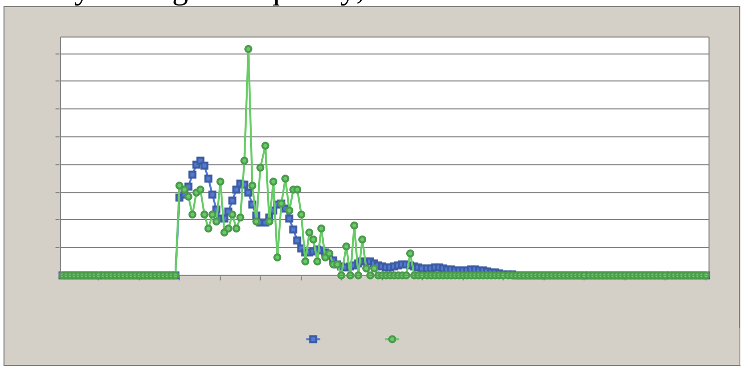
Survey 5 Length Frequency, Year 2001



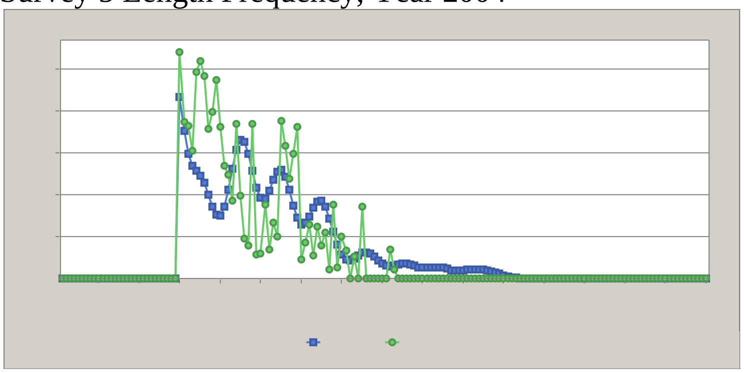
Survey 5 Length Frequency, Year 2002



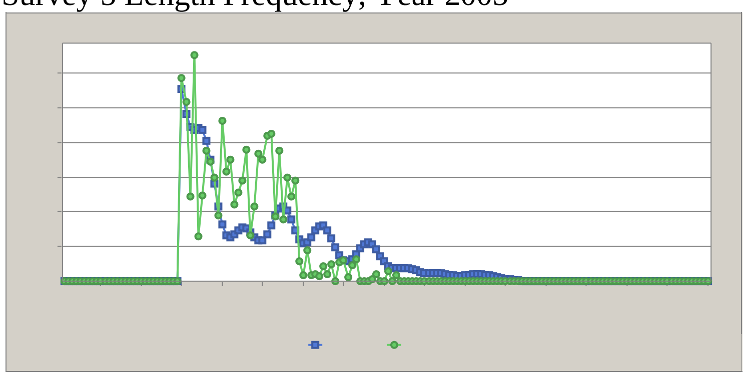
Survey 5 Length Frequency, Year 2003



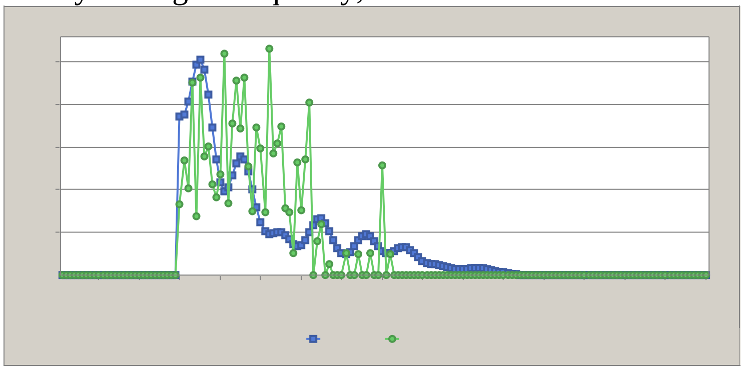
Survey 5 Length Frequency, Year 2004



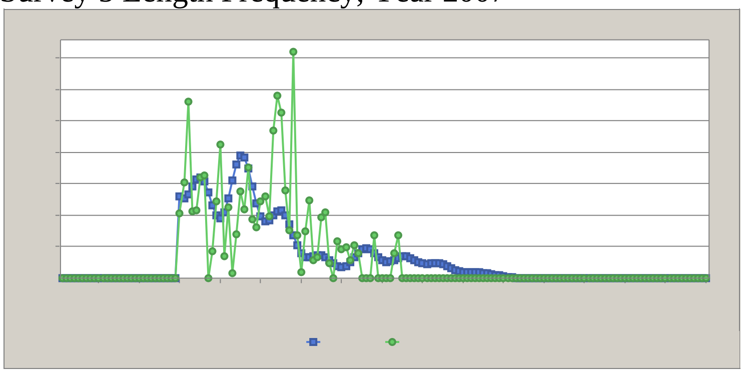
Survey 5 Length Frequency, Year 2005



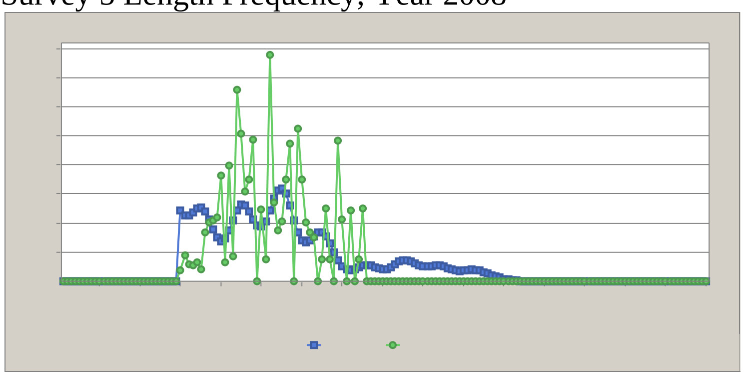
Survey 5 Length Frequency, Year 2006



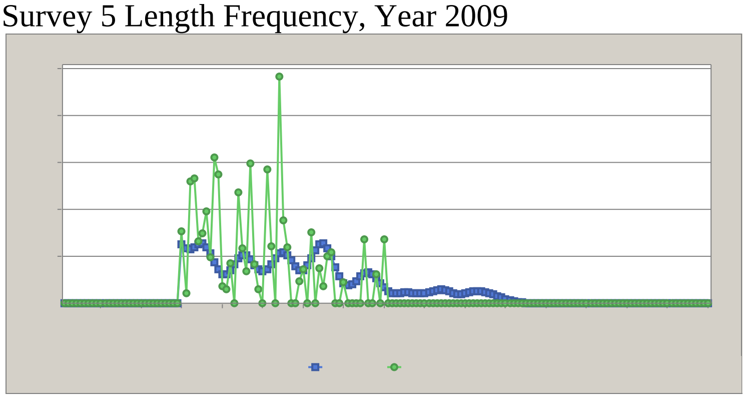
Survey 5 Length Frequency, Year 2007



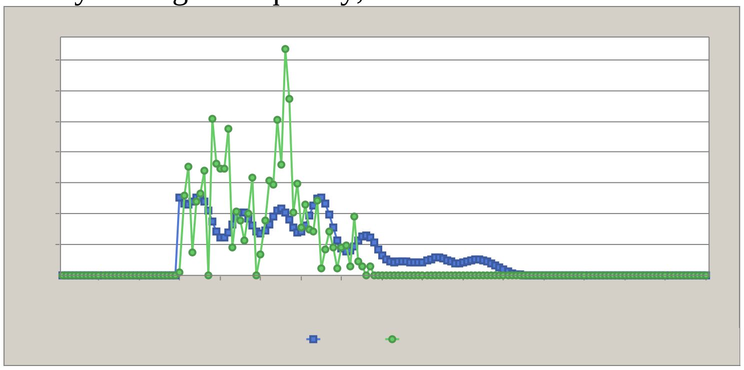
Survey 5 Length Frequency, Year 2008



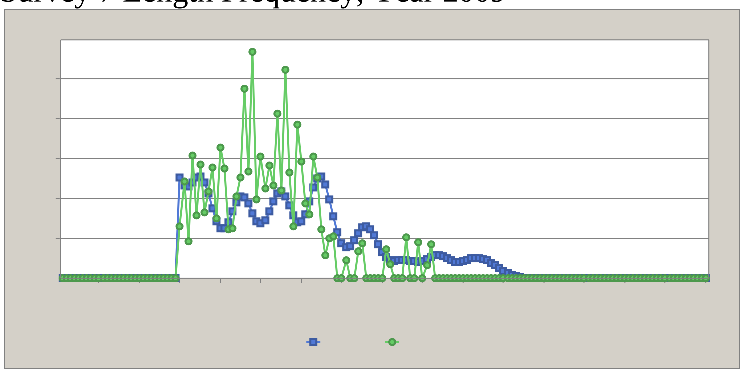
Survey 5 Length Frequency, Year 2009



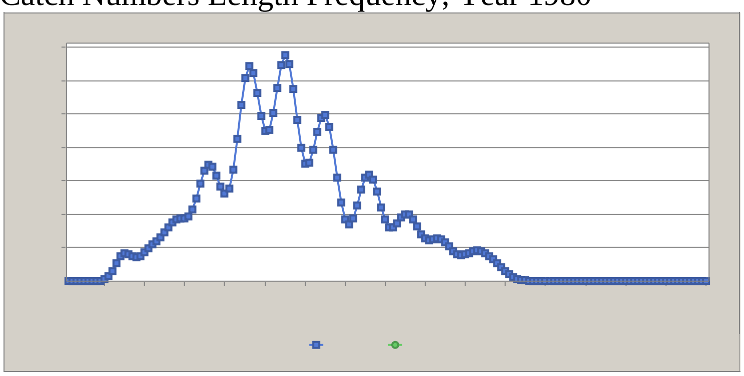
Survey 6 Length Frequency, Year 2009



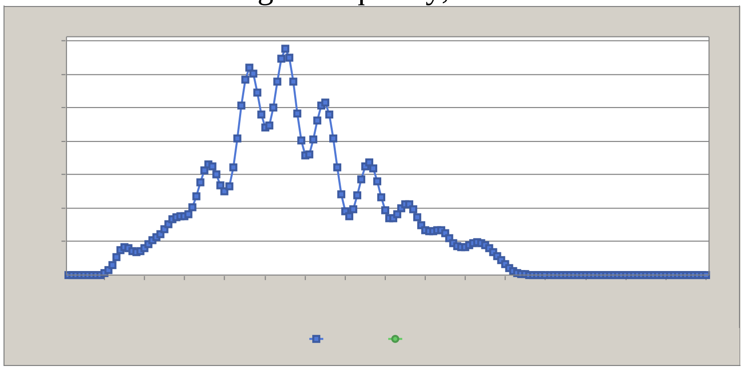
Survey 7 Length Frequency, Year 2009



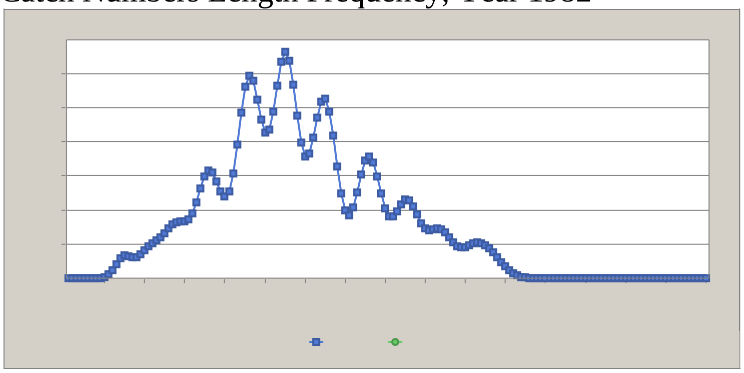
Catch Numbers Length Frequency, Year 1980



Catch Numbers Length Frequency, Year 1981



Catch Numbers Length Frequency, Year 1982



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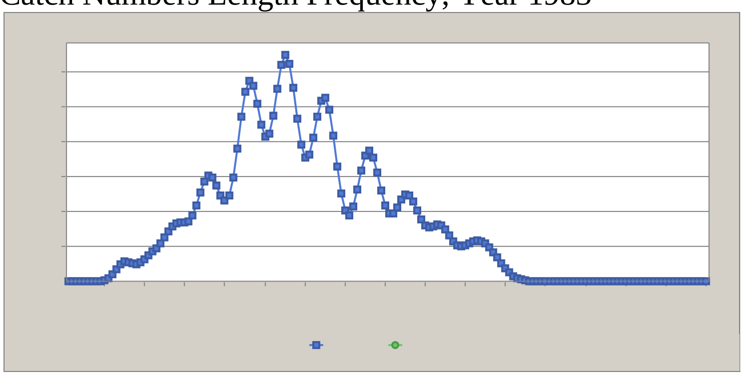

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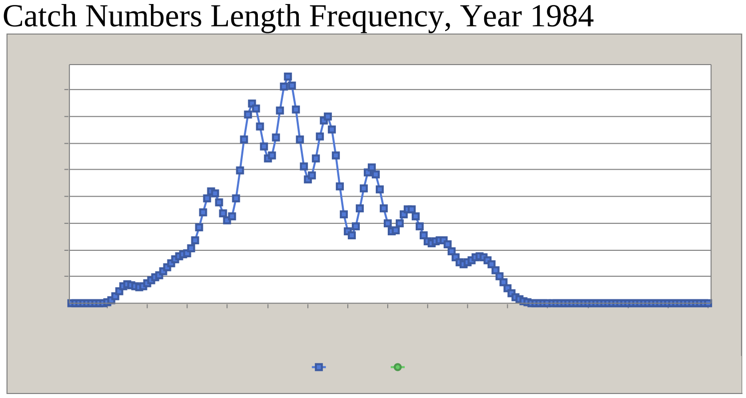

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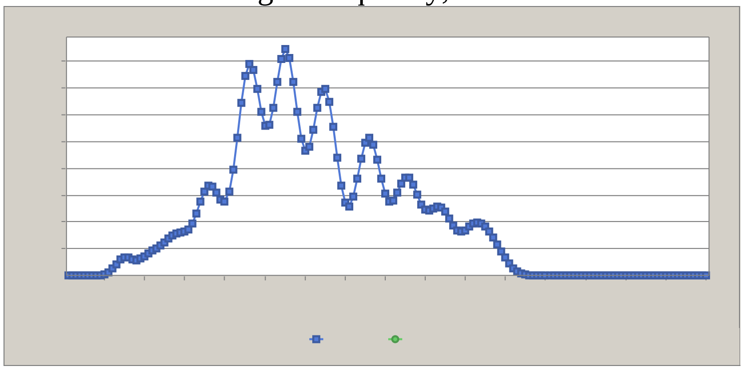
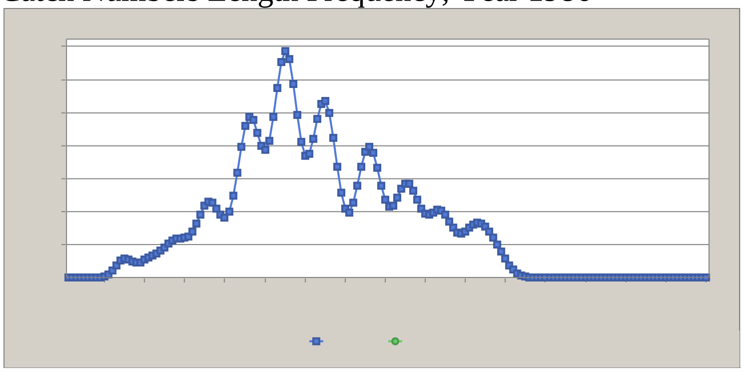
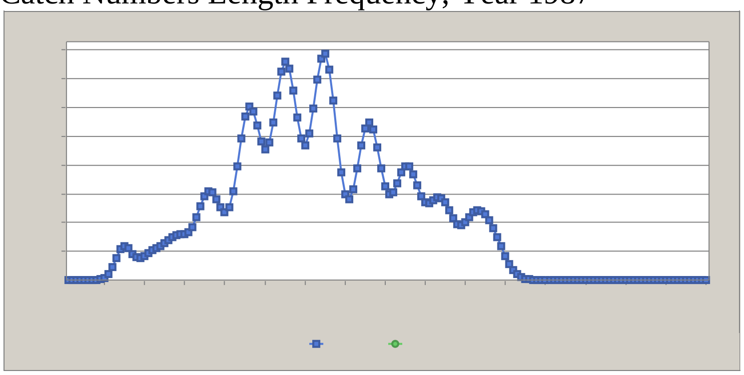


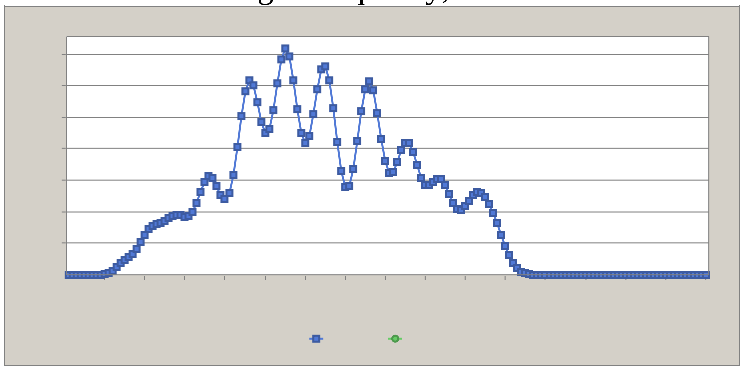
Catch Numbers Length Frequency, Year 1986



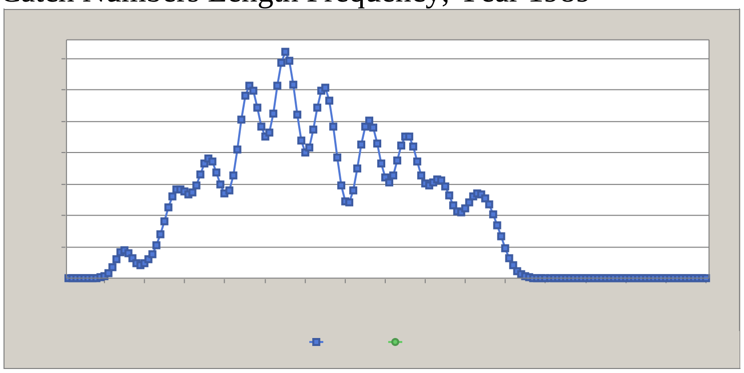
Catch Numbers Length Frequency, Year 1987



Catch Numbers Length Frequency, Year 1988



Catch Numbers Length Frequency, Year 1989



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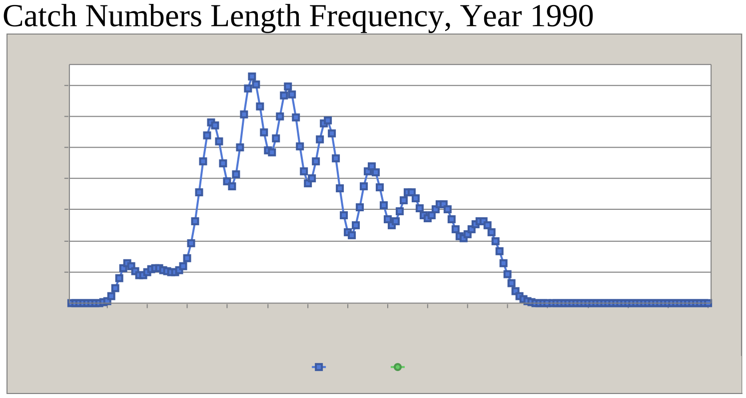
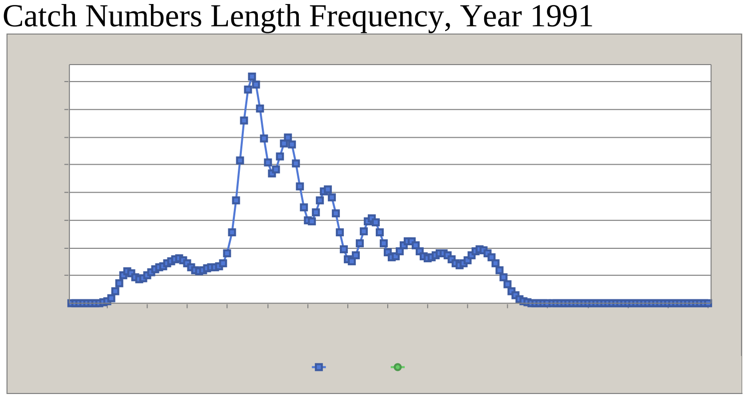
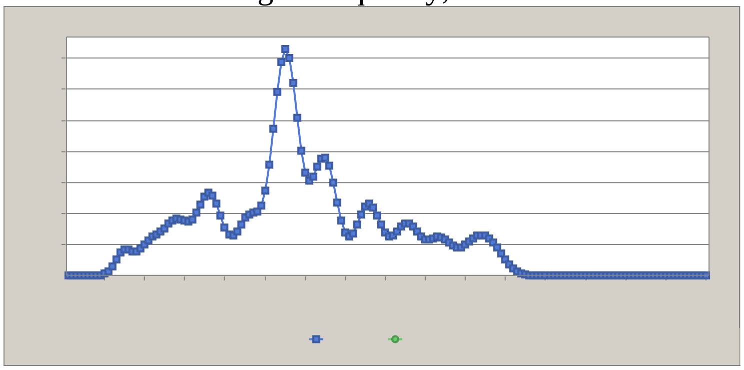


Catch Numbers Length Frequency, Year 1991



Catch Numbers Length Frequency, Year 1992



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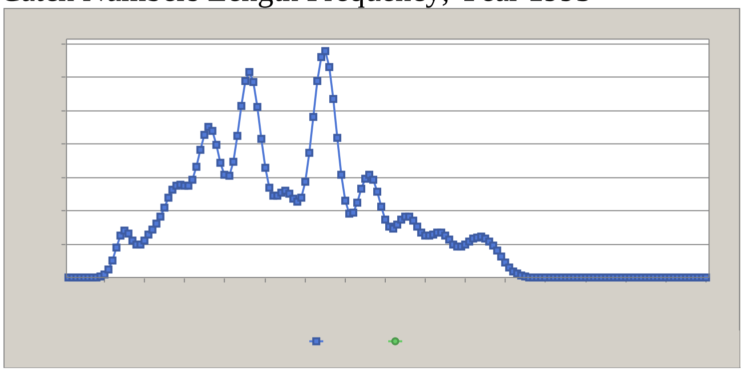

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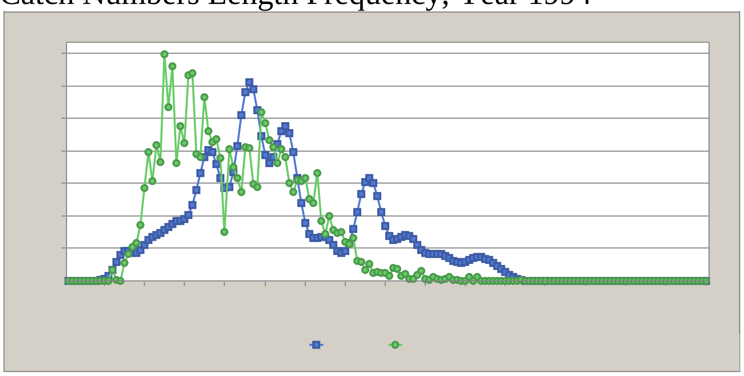

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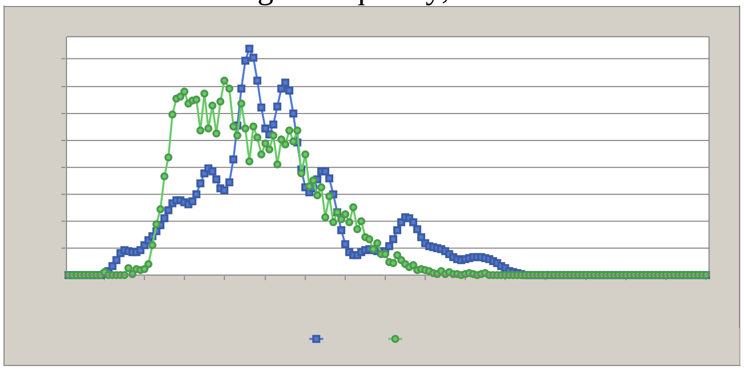
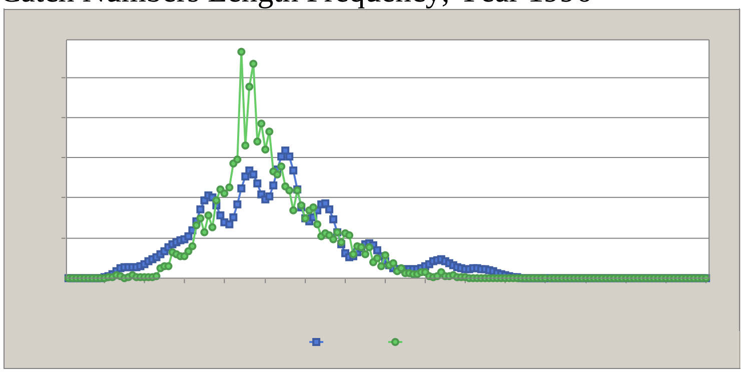


Catch Numbers Length Frequency, Year 1996



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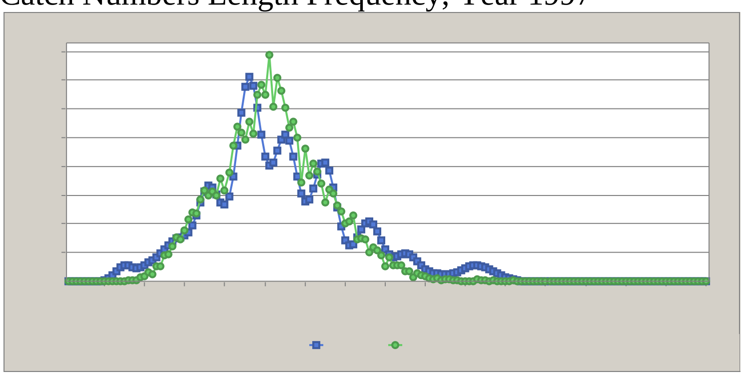
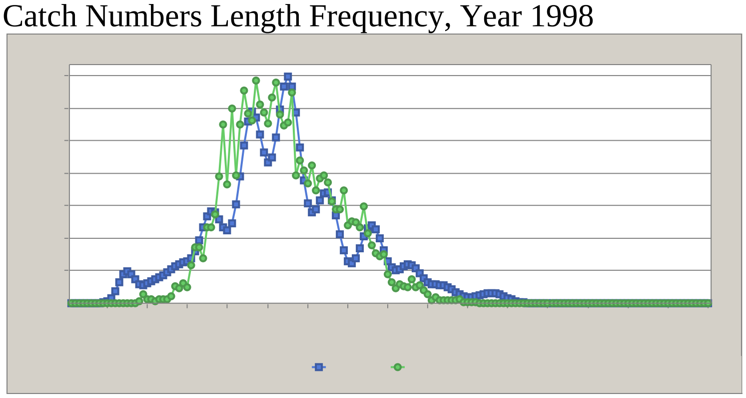
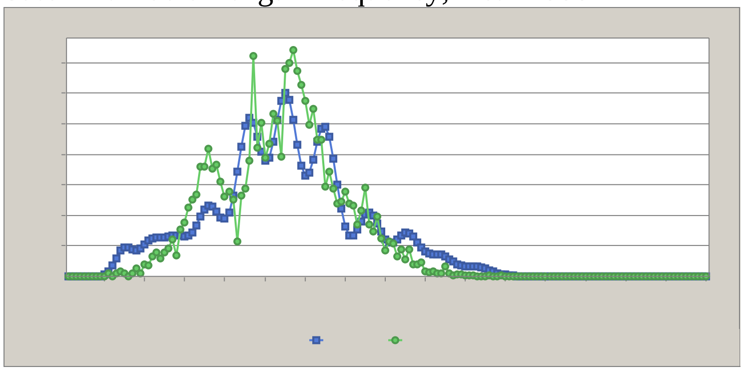


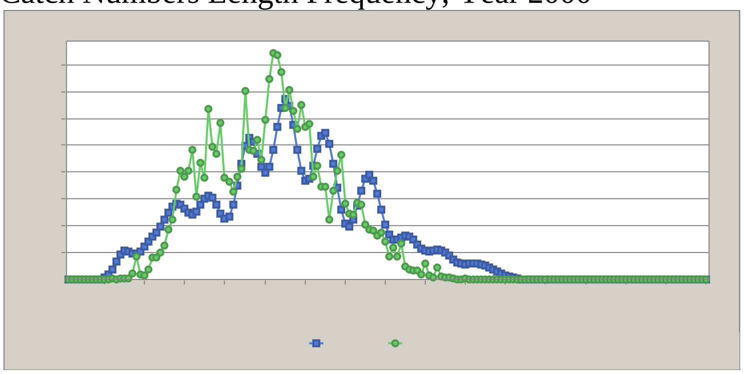
Catch Numbers Length Frequency, Year 1998



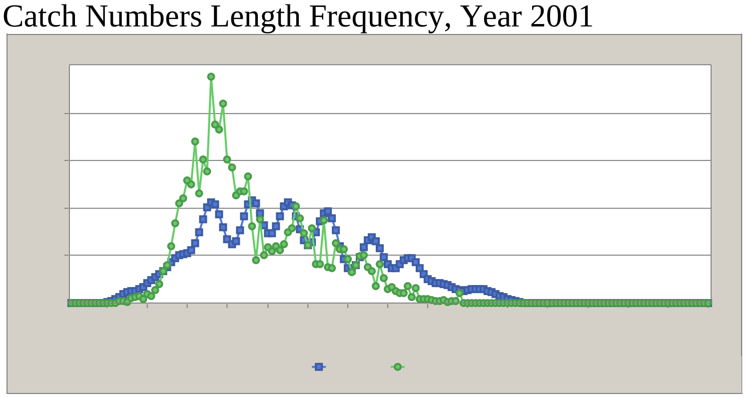
Catch Numbers Length Frequency, Year 1999



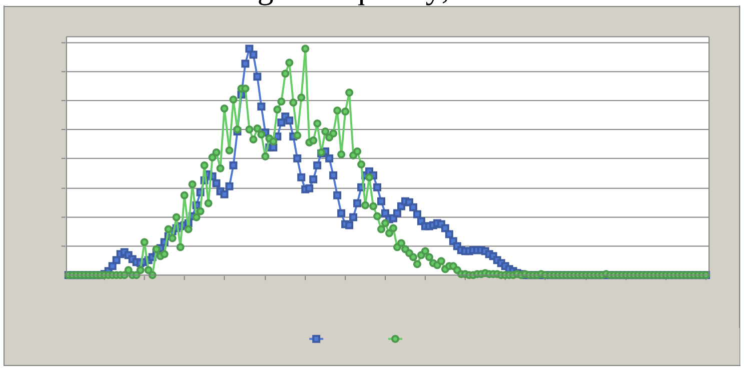
Catch Numbers Length Frequency, Year 2000



Catch Numbers Length Frequency, Year 2001



Catch Numbers Length Frequency, Year 2002



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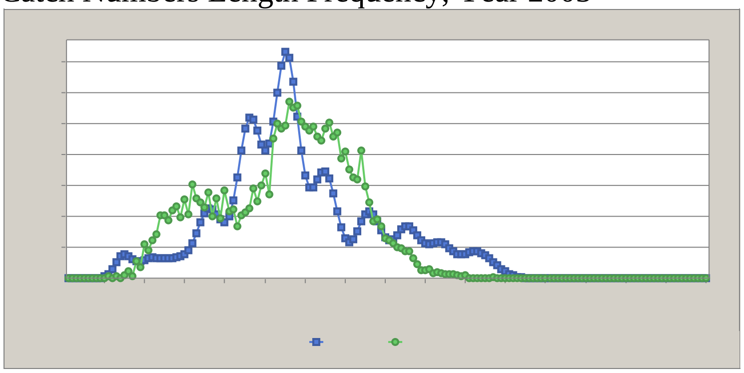

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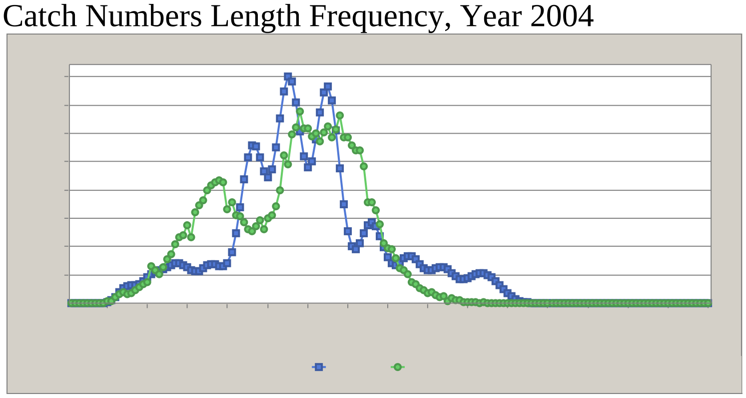
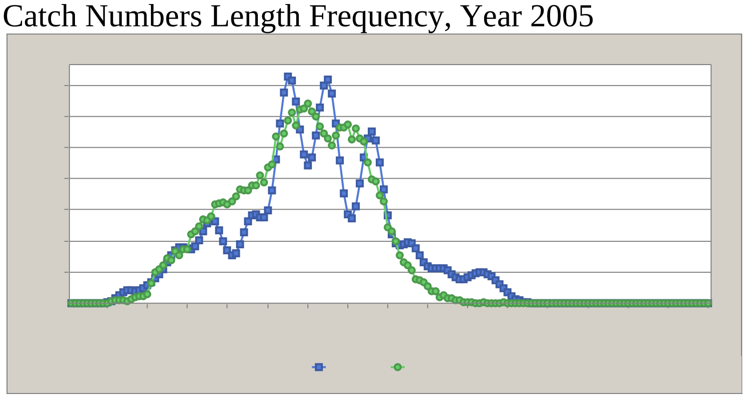
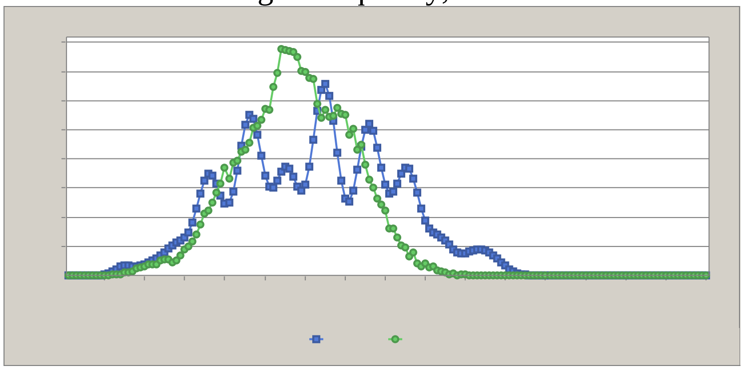


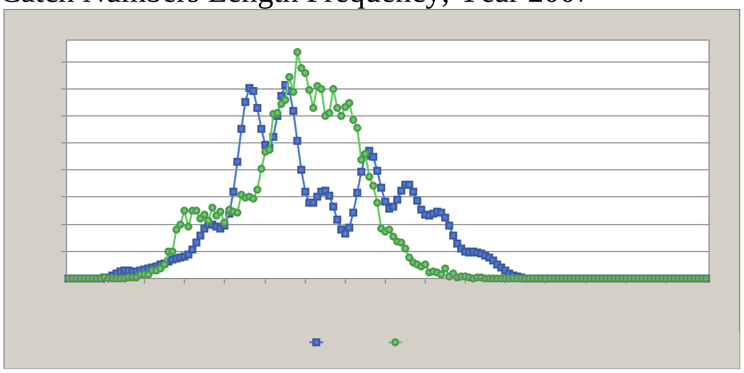
Catch Numbers Length Frequency, Year 2005



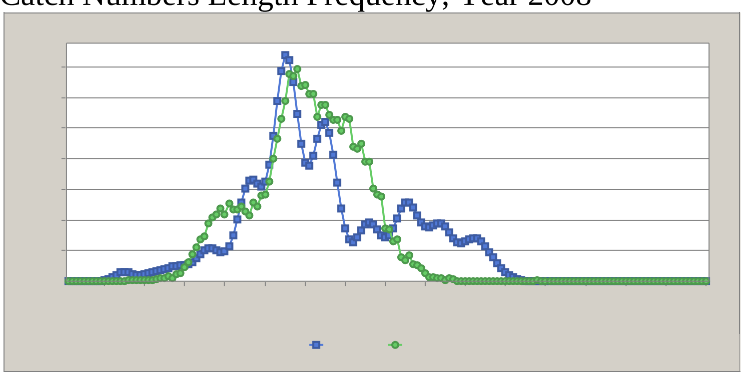
Catch Numbers Length Frequency, Year 2006



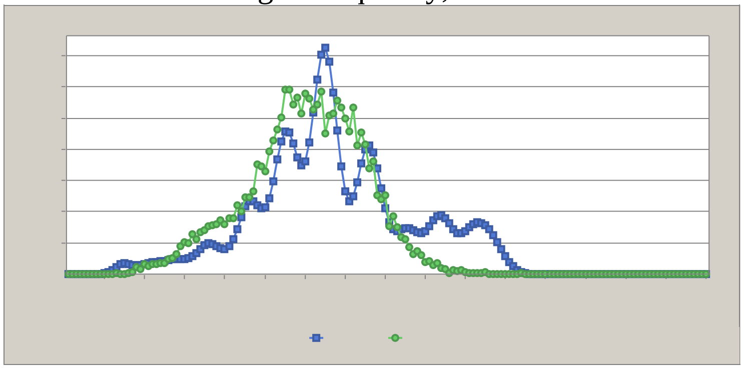
Catch Numbers Length Frequency, Year 2007



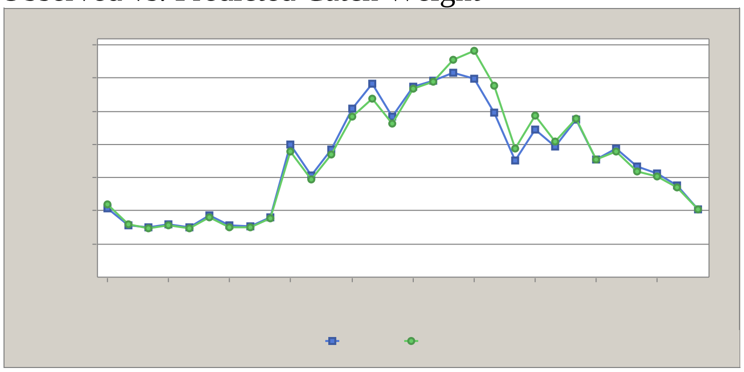
Catch Numbers Length Frequency, Year 2008



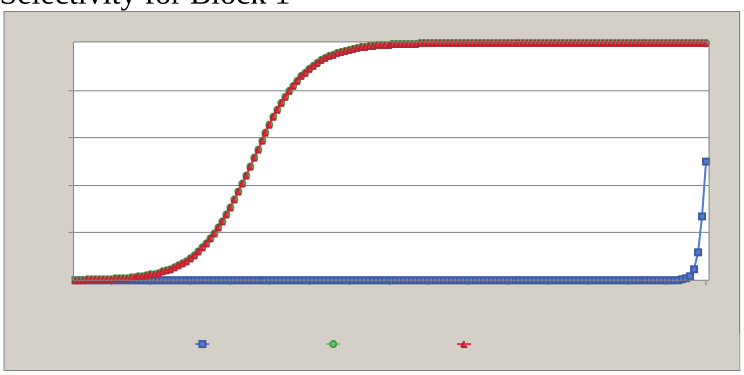
Catch Numbers Length Frequency, Year 2009



Observed vs. Predicted Catch Weight



Selectivity for Block 1



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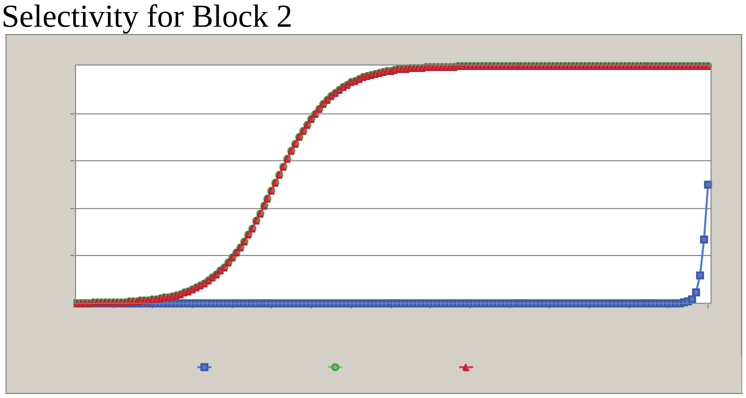
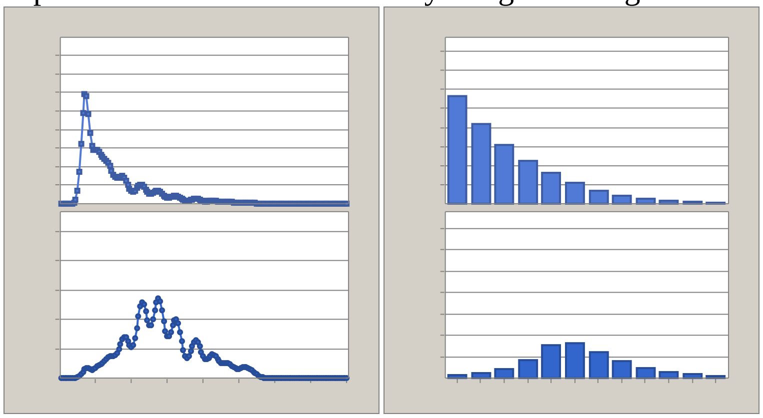
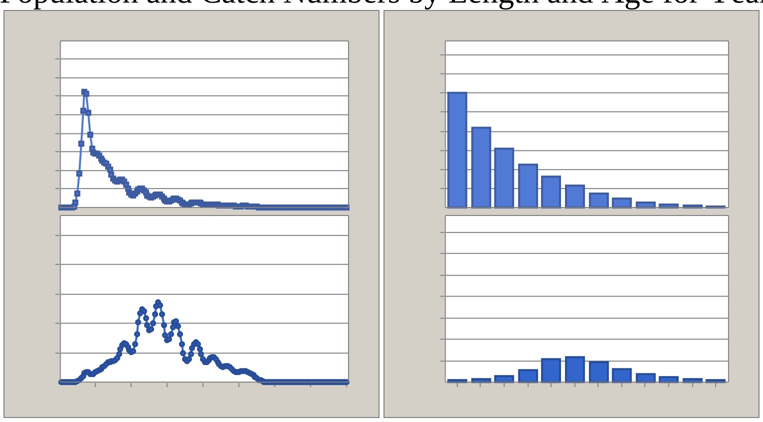


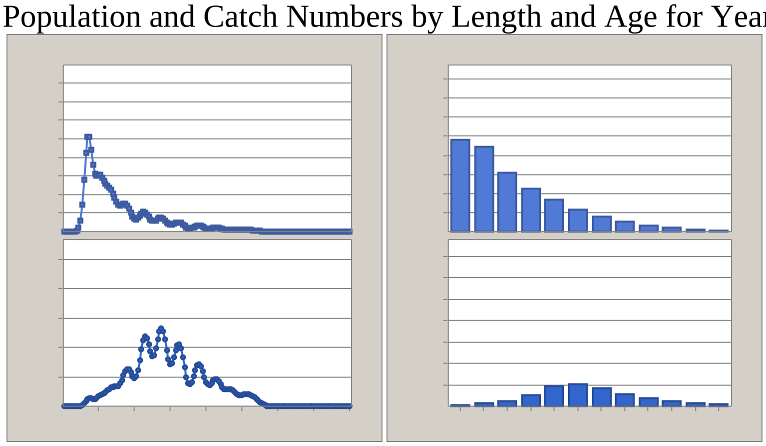
Population and Catch Numbers by Length and Age for Year 1980



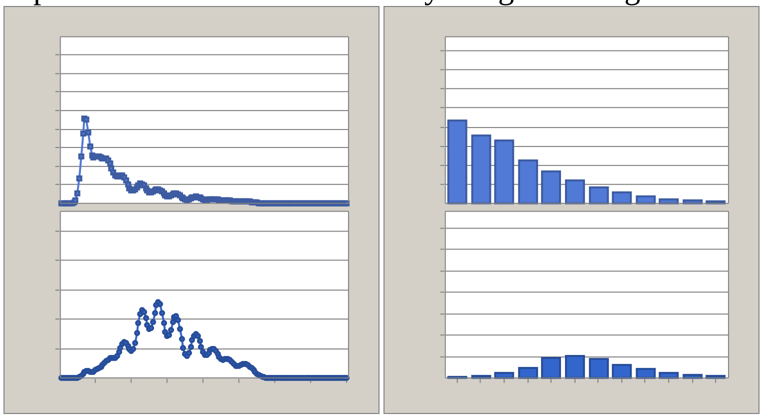
Population and Catch Numbers by Length and Age for Year 1981



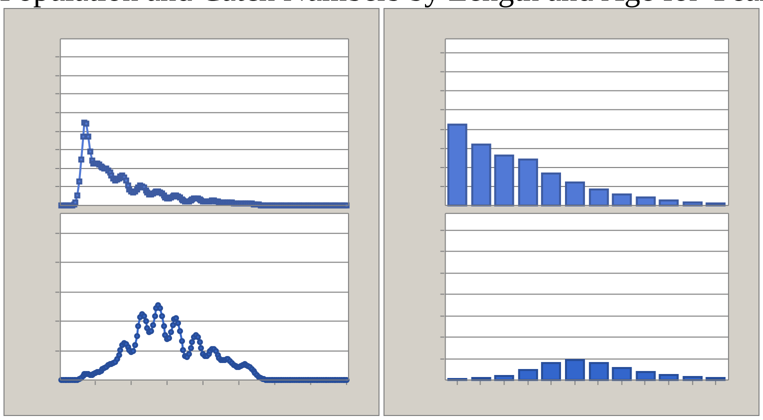
Population and Catch Numbers by Length and Age for Year 1982



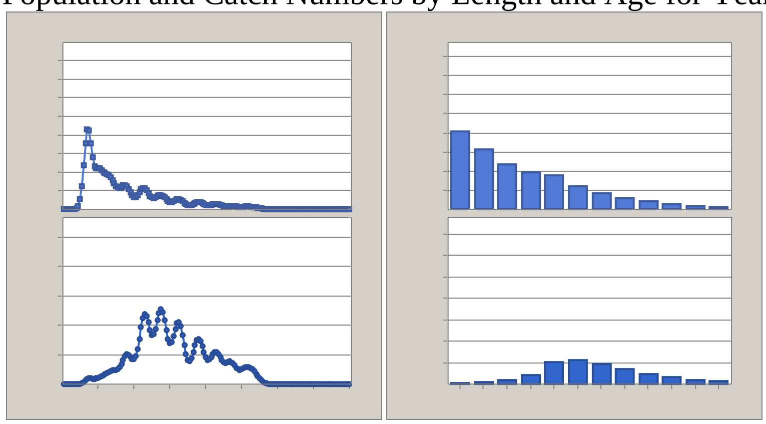
Population and Catch Numbers by Length and Age for Year 1983



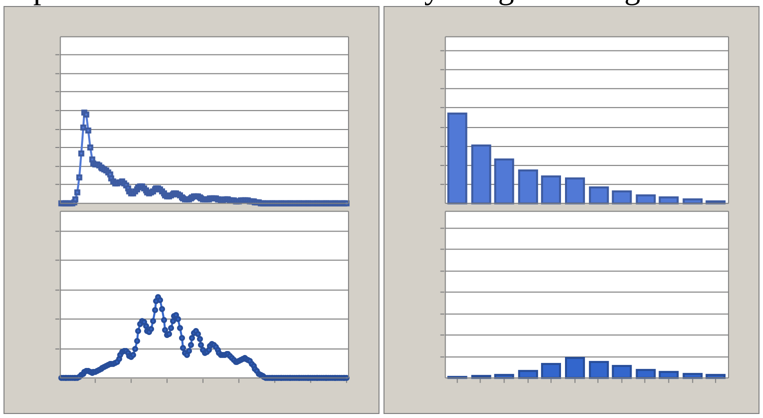
Population and Catch Numbers by Length and Age for Year 1984



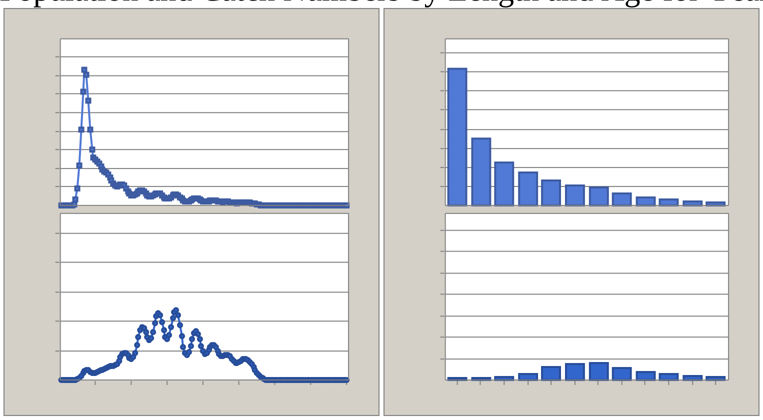
Population and Catch Numbers by Length and Age for Year 1985



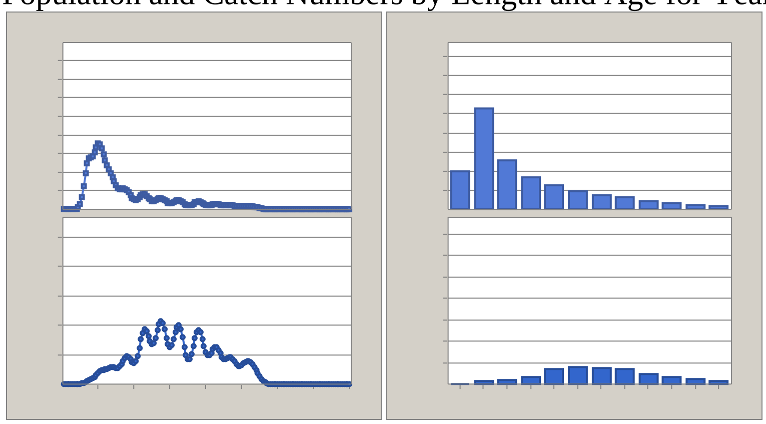
Population and Catch Numbers by Length and Age for Year 1986



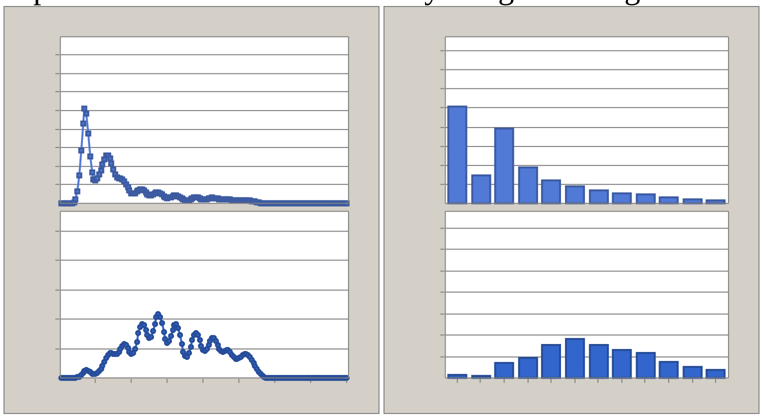
Population and Catch Numbers by Length and Age for Year 1987



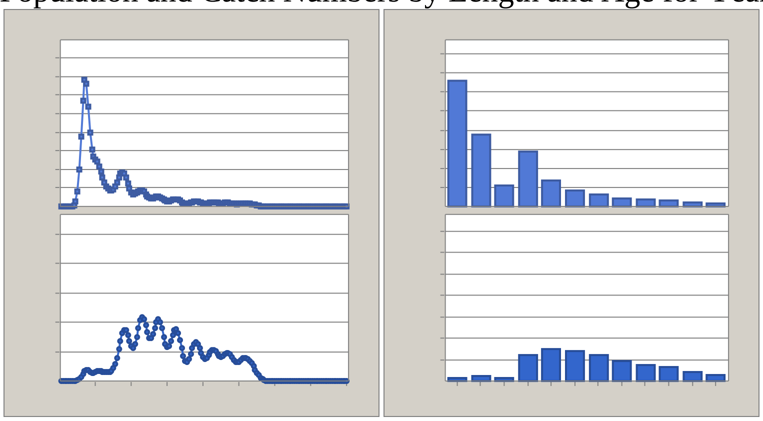
Population and Catch Numbers by Length and Age for Year 1988



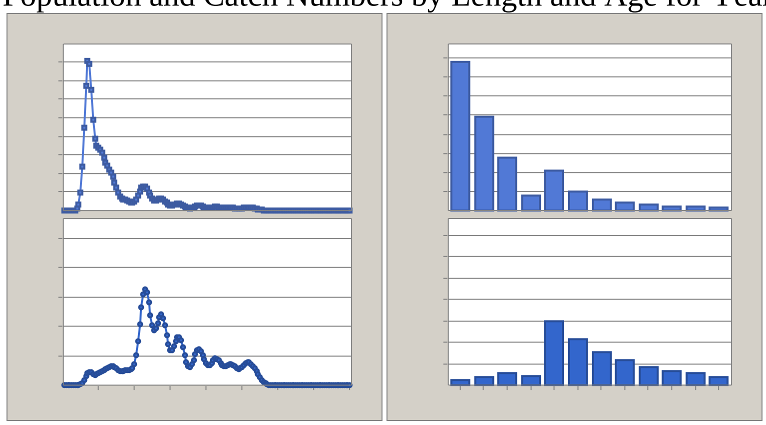
Population and Catch Numbers by Length and Age for Year 1989



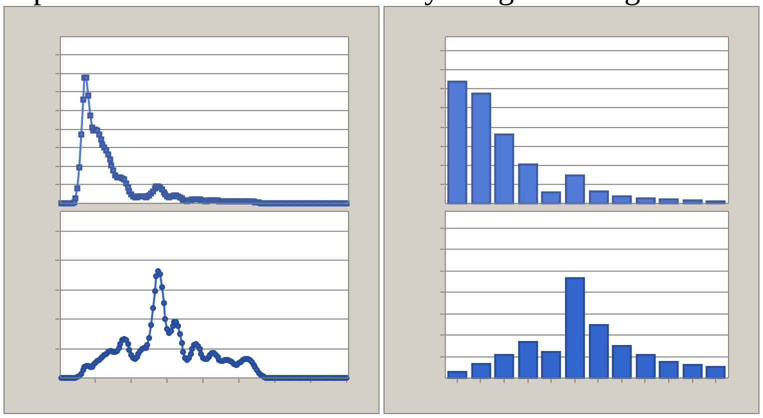
Population and Catch Numbers by Length and Age for Year 1990



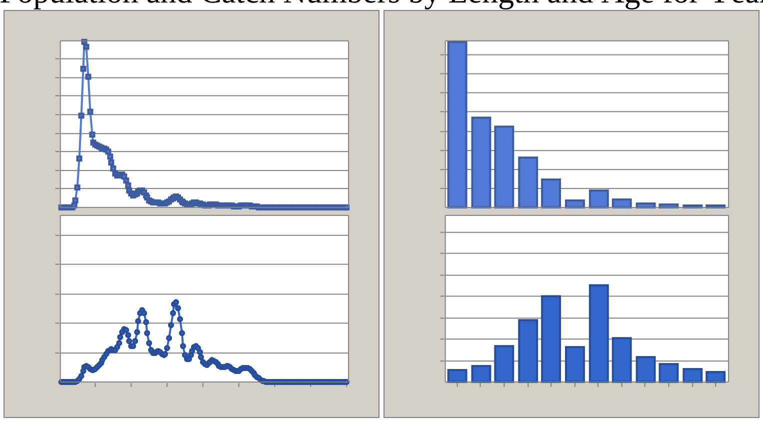
Population and Catch Numbers by Length and Age for Year 1991



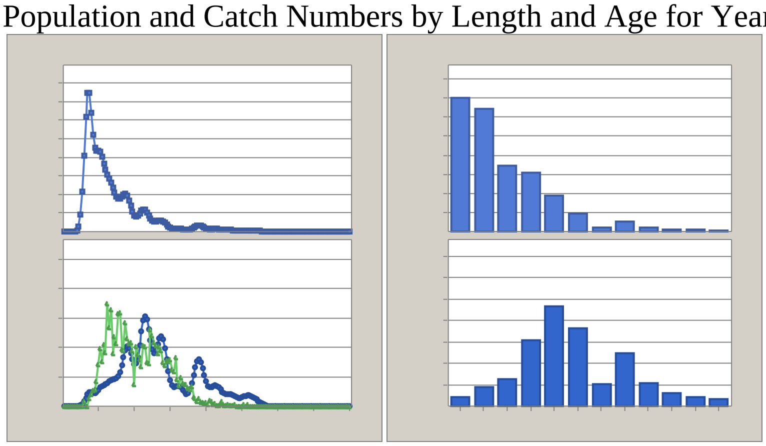
Population and Catch Numbers by Length and Age for Year 1992



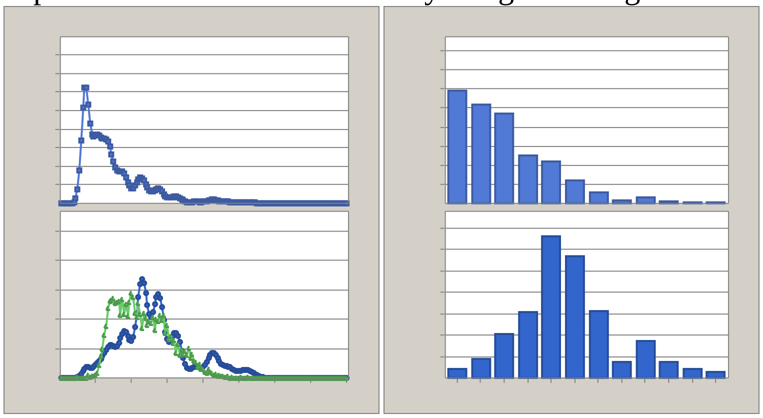
Population and Catch Numbers by Length and Age for Year 1993



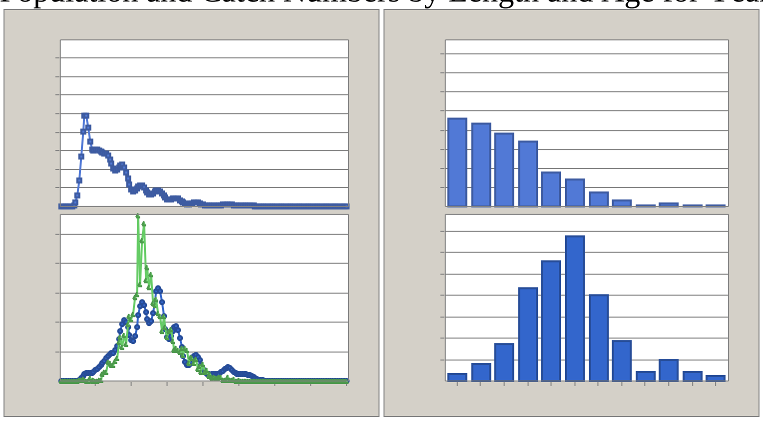
Population and Catch Numbers by Length and Age for Year 1994



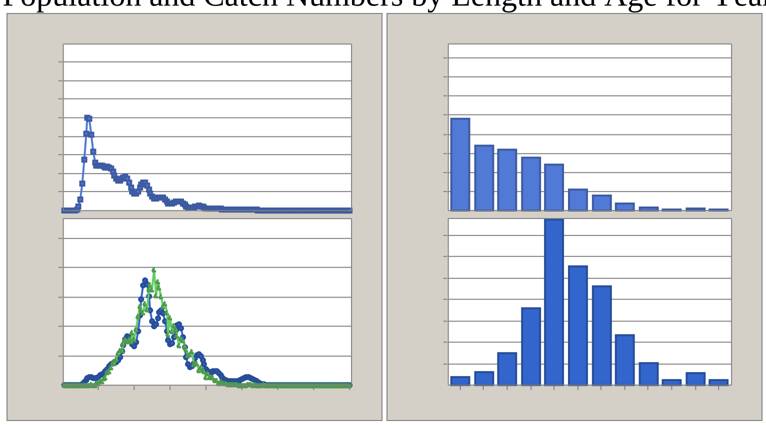
Population and Catch Numbers by Length and Age for Year 1995



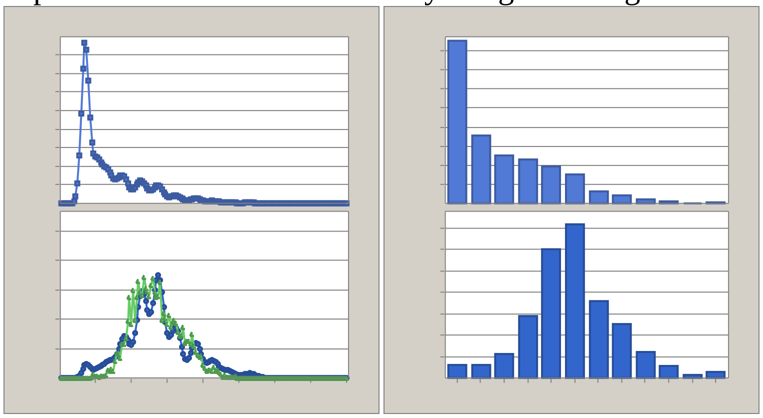
Population and Catch Numbers by Length and Age for Year 1996



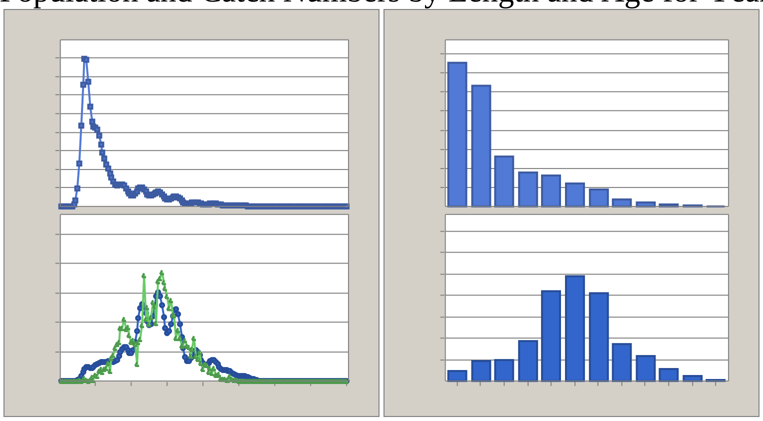
Population and Catch Numbers by Length and Age for Year 1997



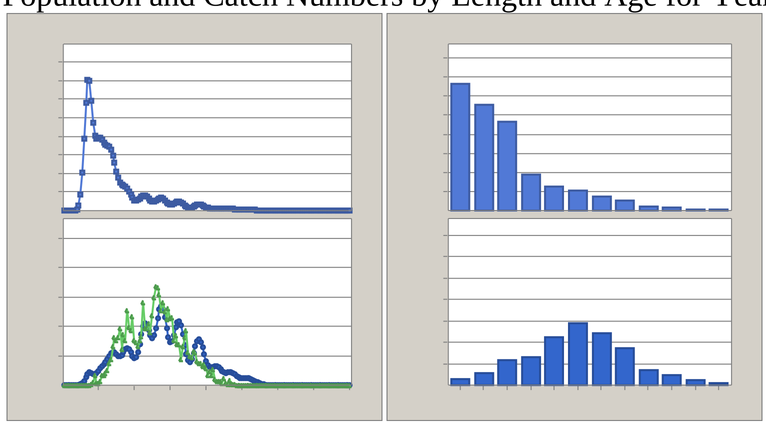
Population and Catch Numbers by Length and Age for Year 1998



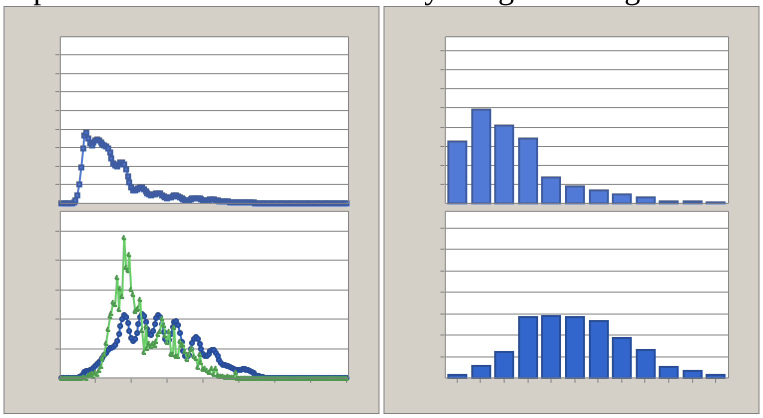
Population and Catch Numbers by Length and Age for Year 1999



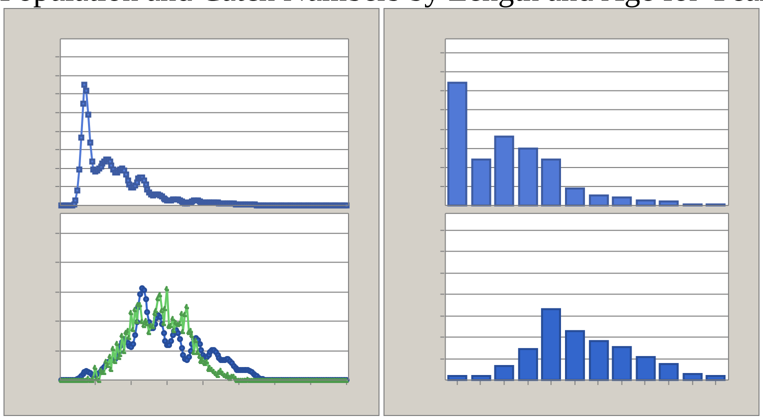
Population and Catch Numbers by Length and Age for Year 2000



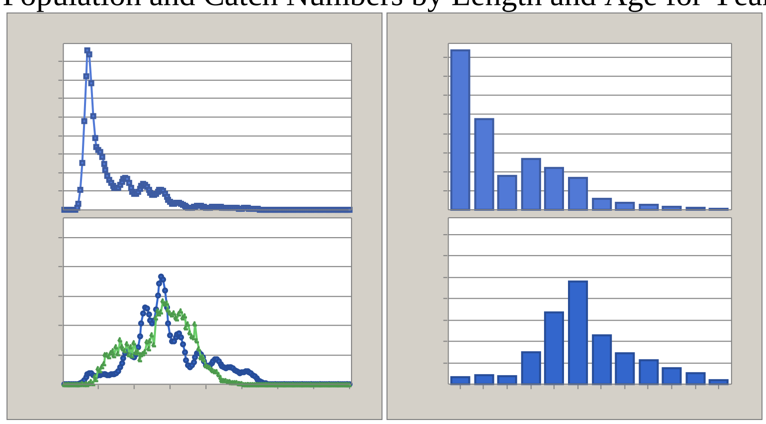
Population and Catch Numbers by Length and Age for Year 2001



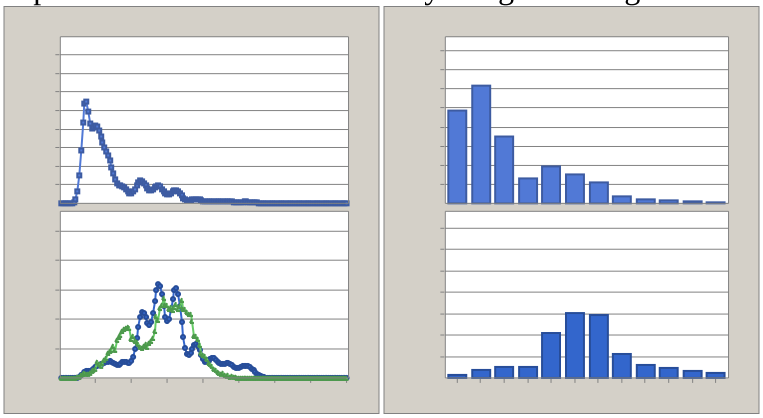
Population and Catch Numbers by Length and Age for Year 2002



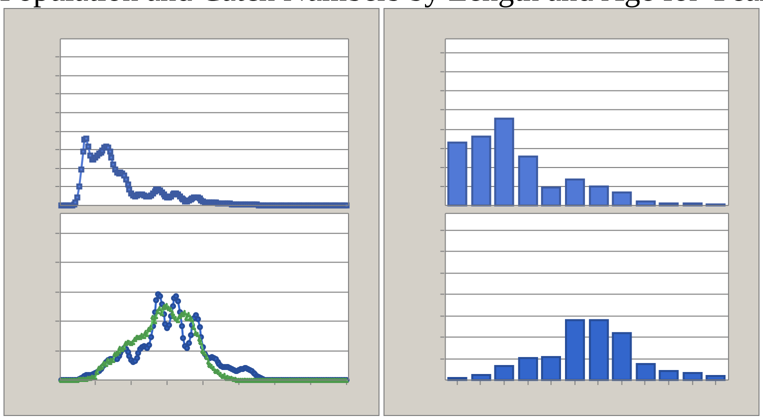
Population and Catch Numbers by Length and Age for Year 2003



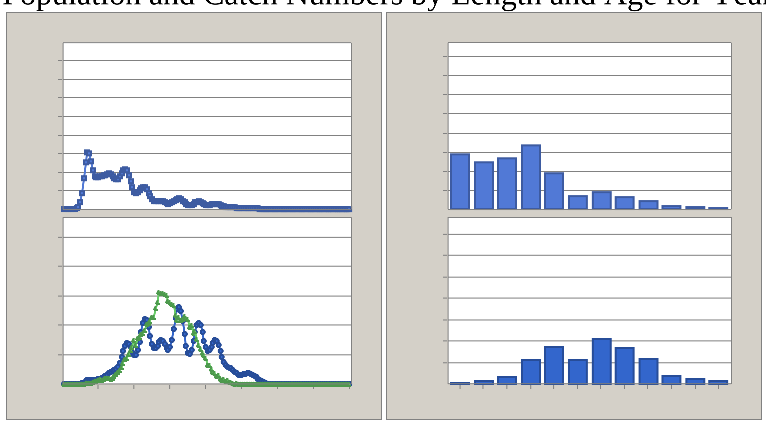
Population and Catch Numbers by Length and Age for Year 2004



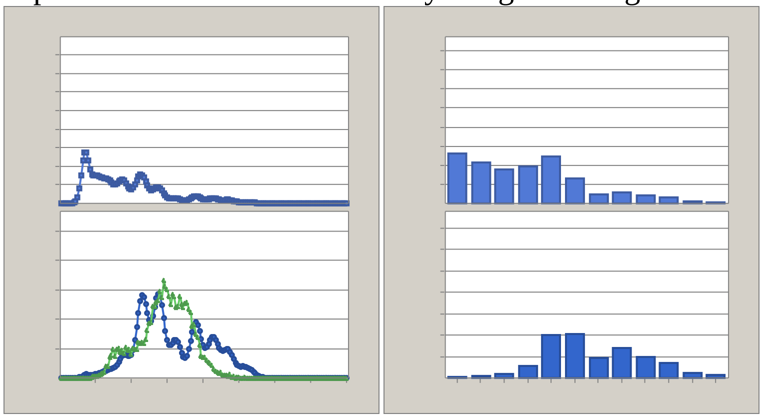
Population and Catch Numbers by Length and Age for Year 2005



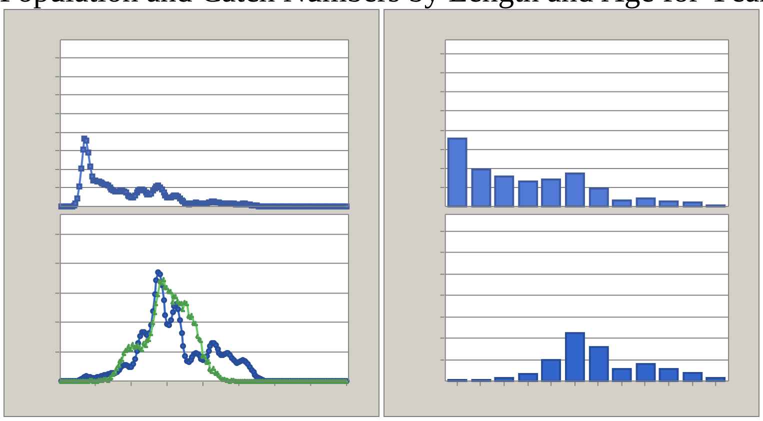
Population and Catch Numbers by Length and Age for Year 2006



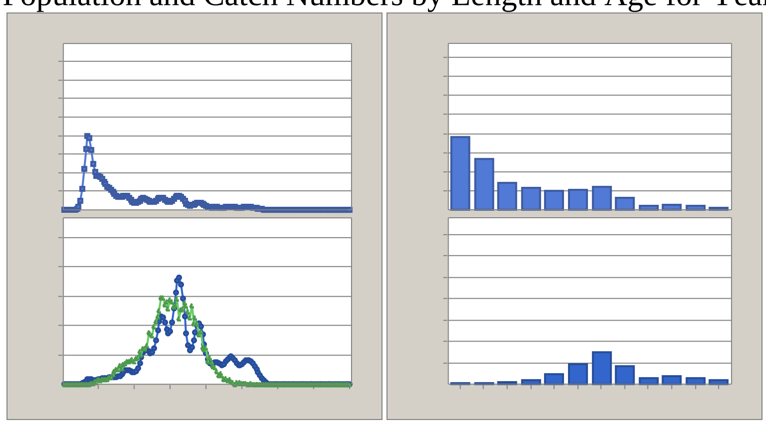
Population and Catch Numbers by Length and Age for Year 2007



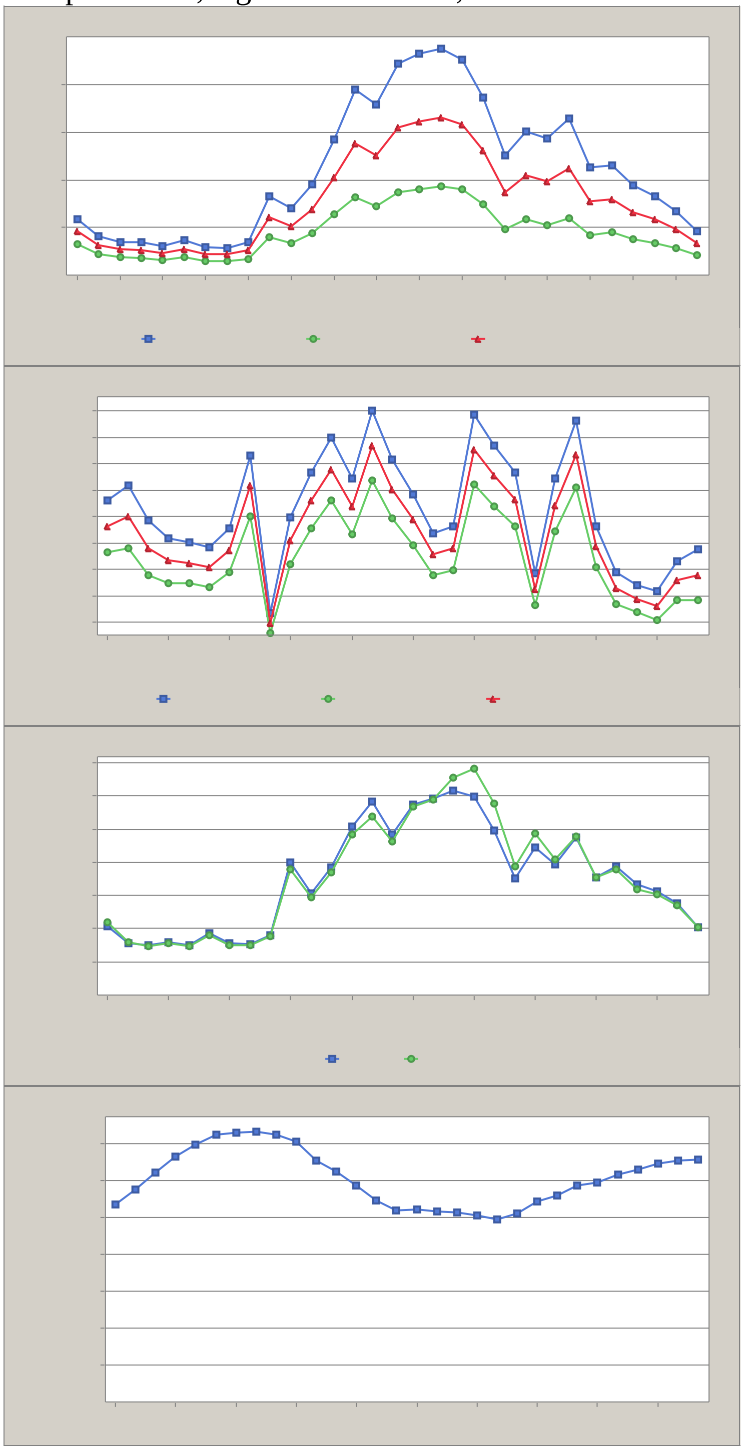
Population and Catch Numbers by Length and Age for Year 2008



Population and Catch Numbers by Length and Age for Year 2009



Grouped Fmult, Age 1 Recruitment, Observed vs. Predicted Catch Weight, and Total Biomass



Appendix A3: A tagging study to assess monkfish (*Lophius americanus*) movements and stock structure in the northeastern United States

Graham Sherwood*, Jonathan Grabowski and Shelly Tallack
Gulf of Maine Research Institute, Portland, Me 04101
*correspondence: gsherwood@gmri.org; (207) 228 1644

A conventional tagging study was conducted to examine movement and mixing rates of monkfish (*Lophius americanus*), respectively, within and between two monkfish management areas in the northeastern United States (the Northern and Southern Management Areas, or NMA and SMA). A total of 2770 monkfish were tagged and released in the autumn of 2007 and winter of 2008 (1006 in the NMA and 1764 in the SMA) and recaptures were monitored over the following 21 months. This study represents the first tagging study for monkfish in the U.S. northeast and almost doubles in effort (i.e., tag releases) the next largest tagging study for *Lophius sp*. The following is a summary of the main findings:

1) The overall reporting rate for filtered recaptures (i.e., days at liberty > 30 days) was 3.2% and this rate was higher in the SMA (3.9%) than in the NMA (1.7%).

2) Tag shedding rate (based on double tagging of all monkfish released), was found to be 18.6% which compares well to shedding rates for other species (e.g., cod).

3) Movements after 30 days at liberty were mostly in the southwest direction and ranged from 1 to 503 km; mean displacement was higher in the NMA than in the SMA: 110.4 ± 129.9 km versus 54.7 ± 58.5 km, and positively correlated with monkfish size in the SMA.

4) Mixing (straying) among management areas was found to be low and unidirectional; no monkfish tagged in the SMA were recaptured in the NMA (although reporting rates were low in the NMA), and we estimate that 9.1% of the monkfish tagged in the NMA moved to the SMA.

5) Growth rate was estimated for a subset of monkfish for which reliable length data existed at the time of recapture (n = 23) to be 10.6 ± 4.7 cm year^{-1} (mean ± std) which compares well with tagging-based estimates of growth for *Lophius piscatorius*. There was a trend (insignificant) for lower growth in larger monkfish which, if coupled to further data and evidence, could call into question the validity of current aging results and the assumption of linear growth in monkfish.

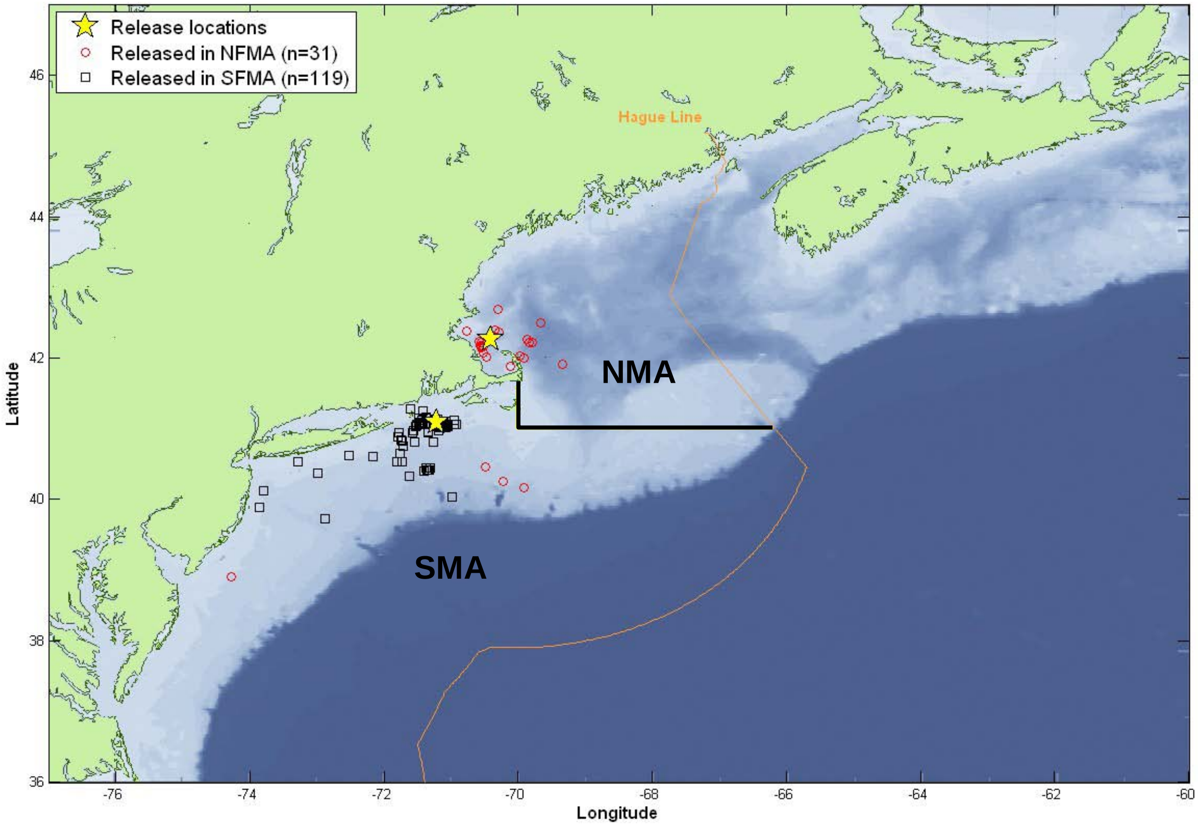


Figure 1. Map showing location of release (note that release sites are close enough within each area to be represented by one star) and recapture locations in the SMA and NMA. Open circles and squares denote the location of fish recapture sites for fish tagged in the NMA and SMA, respectively (see legend). Mean bearing for monkfish released in the NMA was 165°, or almost directly due south (although smaller range movements were to the east). Mean bearing for monkfish released in the SMA was 227° or southwest. Size of released monkfish ranged from 31 to 105 cm (total length).

Table 2. Summary of recaptures by management area and tag color* for non-filtered and filtered (days at liberty > 30 days) data.

			Total recaptures		NMA recaptures		SMA recaptures	
Release Area	Tag color	Releases	Number	Percent	Number	Percent	Number	Percent
No filtering								
NMA	yellow	906	27	3.0	25	2.8	2	0.2
NMA	blue	100	9	9.0	7	7.0	2	2.0
NMA	Total	1006	36	3.6	32	3.2	4	0.4
SMA	yellow	1595	106	6.6	0	0	106	6.6
SMA	blue	169	19	11.2	0	0	19	11.2
SMA	Total	1764	125	7.1	0	0	125	7.1
All	yellow	2501	133	5.3	25	1.0	108	4.3
All	blue	269	28	10.4	7	2.6	21	7.8
All	Total	2770	161	5.8	32	1.2	129	4.7
Filtered for days at liberty > 30								
NMA	yellow	906	13	1.4	11	1.2	2	0.2
NMA	blue	100	8	8.0	6	6.0	2	2.0
NMA	Total	1006	21	2.1	17	1.7	4	0.4
SMA	yellow	1595	59	3.7	0	0	59	3.7
SMA	blue	169	9	5.3	0	0	9	5.3
SMA	Total	1764	68	3.9	0	0	68	3.9
All	yellow	2501	72	2.9	11	0.4	61	2.4
All	blue	269	17	6.3	6	2.2	11	4.1
All	Total	2770	89	3.2	17	0.6	72	2.6

*blue tags were high reward ($100) and yellow tags were standard reward (t-shirt)

B. ATLANTIC SEA SCALLOP STOCK ASSESSMENT FOR 2010

Invertebrate Subcommittee[1]

Terms of Reference

1. Characterize the commercial catch including landings, effort, LPUE and discards. Describe the uncertainty in these sources of data.
2. Characterize the survey data that are being used in the assessment (e.g., regional indices of abundance, recruitment, state surveys, length data, etc.). Describe the uncertainty in these sources of data. Document the transition between the survey vessels and their calibration. If other survey data are used in the assessment, describe those data as they relate to the current assessment (Exclude consideration of future survey designs and methods).
3. Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series, and characterize the uncertainty of those estimates.
4. Update or redefine biological reference points (BRPs; estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, and $F_{MSY;}$ and estimates of their uncertainty). Comment on the scientific adequacy of existing and redefined BRPs.
5. Evaluate stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 4).
6. Develop and apply analytical approaches and data that can be used for conducting single and multi-year stock projections and for computing candidate ABCs (Acceptable Biological Catch; see Appendix to the TORs).
    a. Provide numerical short-term projections (through 2014). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. In carrying out projections, consider a range of assumptions to examine important sources of uncertainty in the assessment.
    b. Comment on which projections seem most realistic, taking into consideration uncertainties in the assessment.
    c. Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC.
7. Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in recent SARC reviewed assessments and review panel reports. Identify new research recommendations.

[1] Meetings and members of the Invertebrate Subcommittee who helped prepare this assessment are listed in Appendix 1.

Executive Summary

TOR 1. *Characterize the commercial catch, effort and CPUE, including descriptions of landings and discards of that species.* (Section 4 and Appendix II)

U.S. sea scallop landings averaged about 26,000 mt meats during 2002-2009, about twice their long-term average. Landings have been particularly high in the Mid-Atlantic Bight region. Fishing effort reached its maximum in 1991, and then declined during the 1990s so that effort in 1999 was less than half that in 1991. Effort in the most recent period has been fairly stable. Landings per unit effort (LPUE) showed general declines from the mid-1960s through the mid-1990s, with brief occasional increases due to strong recruitment. LPUE more than quadrupled between 1998 and 2001, and remained high during 2001-2009. LPUE has been especially high in the Mid-Atlantic and in the Georges Bank access areas (areas that had been closed and are now under special management). Discards of sea scallops were unusually high during 2002-2004, averaging about 10% of landings (by weight), but declined since then, probably due to changes in gear regulations that reduced catches of small individuals.

TOR 2. *Characterize the survey data that are being used in the assessment (e.g., regional indices of abundance, recruitment, state surveys, length data, etc.). Describe the uncertainty in these sources of data. Document the transition between the survey vessels and their calibration. If other survey data are used in the assessment, describe those data as they relate to the current assessment (Exclude consideration of future survey designs and methods).* (Section 5 and Appendices III, IV, V, VI, IX, X, XIV).

Direct and indirect comparisons between the *R/V Albatross IV*, which conducted the NEFSC sea scallop surveys until 2007, and the *R/V Hugh Sharp*, which conducted the 2008-2009 surveys, indicated no statistically significant differences in the catch rates of the two vessels (Appendix IV). However, dredge sensor data indicated that the tow path of the *R/V Hugh Sharp* was about 5% longer than that of the *R/V Albatross IV*, so catches in the time-series were reduced by that amount during 2008-2009.

Comparison of about 140 paired stations between catches of the lined survey dredge and underwater towed camera images (HabCam) gave estimates of survey dredge efficiency of 0.38 in survey strata containing substantial amounts of coarse sediment (gravel, cobble, rock), and 0.44 in all other strata, containing mostly sandy sediments (Appendices IX and X). Edge effects were examined for the SMAST drop camera survey which led to a re-estimation of scallop densities for this survey (Appendix III).

NEFSC sea scallop dredge survey indices were generally low from 1979-1995, and size-frequencies indicated a truncated size distribution with few large scallops. On Georges Bank, abundance and biomass rose substantially in the late 1990s, and then leveled off. After a decline between 2005-2007, indices increased again after strong recruitment was observed during 2007-2009. In the Mid-Atlantic, NEFSC survey indices increased substantially between 1997 and 2003, and have been stable or increased slightly since then. Substantial broadening of the size-structure was observed in both regions starting in the mid-to-late 1990s. SMAST drop video camera survey indices were fairly steady on Georges Bank during 2003-2009. In the Mid-Atlantic, the video estimates declined sharply between 2003 and 2004, and have declined slowly since then. Declines in abundance between 2003 and 2004 were also observed in the lined dredge and the NEFSC winter bottom trawl surveys. These declines are either due to overestimation of the large year class in the 2003 survey indices or high natural or fishing induced mortality of this year class or some combination of these effects.

TOR 3. *Estimate fishing mortality, spawning stock biomass, and total stock biomass for the current year and characterize the uncertainty of those estimates. If possible, also include estimates for earlier years.* (Section 6 and Appendix XI).

A dynamic size-based stock assessment model (CASA) was used to estimate biomass, abundance and fishing mortality. This model was introduced in a preliminary version in NEFSC (2004) and used as the primary assessment model in NEFSC (2007). Data used in CASA included commercial catch, LPUE, and fishery shell height compositions, the NEFSC sea scallop and winter trawl surveys, the SMAST large camera video survey, growth increment data from scallop shells, and shell height/meat weight data adjusted to take into account commercial practices and seasonality. Because both the video and lined dredge survey (via the paired dredge/camera experiment) give estimates of scale (absolute abundances), prior estimates for efficiencies of these two surveys were used in the CASA model.

The sea scallop stock was assessed in two components (Georges Bank and Mid-Atlantic) separately and then combined. Estimates of fishing mortality were made from 1975-2009 for both regions. The models generally gave good fits to survey and commercial data, but there was tension in the Mid-Atlantic Bight model between the efficiency priors (especially for the video survey) and the recent stable or declining trends observed in surveys. Possible mild retrospective patterns were observed in the model, especially in the Mid-Atlantic Bight.

Model output and fishery size composition data indicate a substantial shift in selectivity towards larger scallops Fishing mortality rates in 2009 are comparable to revised reference points but they are not comparable among fishery selectivity periods except as measures of fishing mortality on the fully selected individuals because of the shifts in selectivity. Whole stock fully recruited fishing mortality increased from 1975-1992, reaching a peak of 1.47 in 1992, rapidly declined during the late 1990s, and has been fairly stable since 2002. Estimated fishing mortality in 2009 was 0.18 (Georges Bank), 0.60 (Mid-Atlantic) and 0.378 for the whole stock.

Combined model estimated abundances and biomass increased rapidly in the decade starting in 1994, and have been stable or slightly increasing since then. July 1, 2009 estimated biomasses were 62,470 mt meats for Georges Bank and 67,233 mt meats in the Mid-Atlantic. Whole stock abundance and biomass estimates for July 1 2009 were 4,446 billion scallops and 129,703 mt meats. Both abundance and biomass for 2009 were at the maximum of the 1975-2009 time series.

TOR 4. *Update or redefine biological reference points (BRPs; estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, and $F_{MSY;}$ and estimates of their uncertainty). Comment on the scientific adequacy of existing and redefined BRPs.* (Section 7).

The per recruit reference points F_{MAX} and B_{MAX} had been used as proxies for F_{MSY} and B_{MSY} in previous assessments. NEFSC (2007) estimated $F_{MAX} = 0.29$ and $B_{MAX} = 109{,}000$ mt meats (January 1 biomass). These estimates were updated in this assessment using new data and the current CASA model: $F_{MAX} = 0.30$ and $B_{MAX} = 125{,}000$ mt meats, based on January 1 biomass as was used in NEFSC (2007).

During the last benchmark assessment (NEFSC 2007), it was recommended that alternative reference points be explored because the changes in selectivity have made yield per recruit curves increasingly flat, which makes F_{MAX} more difficult to estimate and sensitive to small changes in assumed parameters.

A new method for estimating reference points is proposed in this assessment (SYM – Stochastic Yield Model) which explicitly takes into account uncertainties in per recruit and stock-recruit relationships to estimate F_{MSY} and B_{MSY} using Monte-Carlo simulations. This model estimated whole-stock F_{MSY} = 0.38, B_{MSY} = 125,358 mt meats (July 1 biomass), and MSY = 24,975 mt meats. This assessment used July 1 model biomass since it is a more representative of the actual biomass in the population. July 1 model abundance and biomass are always lower than those on January 1 because all growth and recruitment in the model occur on January 1.

TOR 5. *Evaluate stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 4).* (Section 8).

According to the Amendment 10 overfishing definition (NEFMC 2003), sea scallops are overfished when the survey biomass index for the whole stock falls below 1/2 B_{TARGET}. The target biomass estimated in NEFSC (2007), B_{TARGET} = 109,000 mt on January 1, was calculated as the median recruitment in the survey time series times BPR_{MAX}, the biomass per recruit obtained when fishing at F_{MAX}. NEFSC (2007) estimated F_{MAX}= 0.29, which has been used since then as the overfishing threshold. The updated values in this assessment are F_{MAX} = 0.30 and B_{MAX} = 85,000 mt (July 1 biomass). The new proposed stochastic MSY reference points are F_{MSY} = 0.38 and B_{MSY} = 125,358 mt (July 1).

Estimated whole-stock biomass in for January 1, 2009 was 158,610 mt meats, and 129,703 mt for July 1. These estimates are above the biomass target of 109,000 mt meats from NEFSC (2007) as well as the new biomass targets (85,000 mt meats July 1 using per recruit analysis, 125,358 mt meats using the stochastic yield approach). Thus, the current estimated biomass is more than twice the biomass threshold of 1/2 B_{TARGET}, regardless of which reference point approach is used. The sea scallop stock was therefore not overfished in 2009.

Estimated whole stock fishing mortality was 0.38 for the whole stock (to three decimal places 0.378), which is above the NEFSC (2007) overfishing threshold of 0.29 and its updated value of 0.30, but equal to the proposed estimate of F_{MSY} = 0.38. Therefore, overfishing was not occurring in 2009 based on the new recommended overfishing definition; however, overfishing would be occurring if the previous definition or its updated value had been used.

TOR 6. *Develop and apply analytical approaches and data that can be used for conducting single and multi-year stock projections and for computing candidate ABCs (Acceptable Biological Catch; see Appendix to the TORs).*

a.Provide numerical short-term projections (through 2014). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. In carrying out projections, consider a range of assumptions to examine important sources of uncertainty in the assessment.

b.Comment on which projections seem most realistic, taking into consideration uncertainties in the assessment.

c.Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC. (Section 8)

The recommended projection model is spatially explicit and accommodates differences among regions in recruitment, growth, initial size structure, shell height/meat weight relationships, management approach (open vs. closed areas and catch quota vs. limits on fishing effort), intensity of fishing effort, and other factors. Projections done assuming status-quo management but varying initial conditions, natural mortality and recruitment indicate that biomass and landings are expected to increase modestly until 2012, and then level off. There is less than a 0.1% chance of the stock becoming overfished by 2014. The stock has low vulnerability to becoming overfished.

TOR 7. *Review, evaluate and report on the status of the SARC/Working Group Research Recommendations offered in recent SARC reviewed assessments.* Completed (Section 9)

Progress has been made on some of the recommendations, such as estimation of natural mortality and seasonal growth models. But no progress has been made on others, such as obtaining better estimates of discard and incidental mortality.

Introduction

Life History

The Atlantic sea scallop, *Placopecten magellanicus*, is a bivalve mollusk that occurs on the eastern North American continental shelf north of Cape Hatteras. Major aggregations in US waters occur in the Mid-Atlantic from Virginia to Long Island, on Georges Bank, in the Great South Channel, and in the Gulf of Maine (Hart and Chute 2004). In Georges Bank and the Mid-Atlantic, sea scallops are harvested primarily at depths of 30 to 100 m, whereas the bulk of landings from the Gulf of Maine are from near-shore waters. This assessment focuses on the two main portions of the sea scallop stock and fishery, Georges Bank in the north and the Mid-Atlantic in the south (Figure B-1). Results for Georges Bank and the Mid-Atlantic are combined to evaluate the stock as a whole. Assessments of the Gulf of Maine populations can be found in Appendices V and VI.

US landings during 2003-2009 exceeded 24,000 mt (meats) each year, roughly twice the long-term mean.[2] US ex-vessel sea scallop revenues during 2005-2009 averaged $389 million, making it the most valuable US fishery during this time. Unusually strong recruitment in the Mid-Atlantic Bight area and increased yield per recruit due to effort reduction and fishing gear modification measures are the key reasons for high recent landings. The mean meat weight of a landed scallop during 2005-2009 was over 25 g, compared to less than 14 g during the early to mid 1990s.

Area closures and reopenings have a strong influence on sea scallop population dynamics (Figure B-1). Roughly one-half of the productive scallop grounds on Georges Bank and Nantucket Shoals were closed to both groundfish and scallop gear during most of the time since December 1994. Limited openings to allow scallop fishing in closed areas contributed more than half of Georges Bank landings during 1999-2000 and since 2004.

In the Mid-Atlantic, there have been five rotational scallop closures. Two areas (Hudson Canyon South and Virginia Beach) were closed in 1998 and then reopened in 2001. Although the small Virginia Beach closure was unsuccessful, scallop biomass built up in Hudson Canyon Closed Area while it was closed, and substantial landings were obtained from Hudson Canyon during 2001-2007. This area was again closed in 2008, and will likely reopen in 2011. A third rotational closure, the Elephant Trunk area east of Delaware Bay, was closed in 2004, after extremely high densities of small scallops were observed in surveys during 2002 and 2003. About 30,000 mt of scallops have been landed from that area since it reopened in 2007. A fourth closed area (Delmarva), directly south of the Elephant Trunk area, was closed in 2007 and was reopened in 2009.

[2] In this assessment, landings and biomass figures are metric tons (mt) of scallop meats, unless otherwise indicated.

Early attempts to model sea scallop population dynamics (NEFSC 1992, 1995, 1997, 1999) were not successful because biomass estimates were less than the minimum swept area biomass obtained from the NEFSC scallop survey (NEFSC 1999). In lieu of model based estimates, fishing mortality was estimated in NEFSC (1999, 2001 and 2004) using a simple rescaled *F* method which relies heavily on survey and landings data. A size-structured forward projecting model (CASA, based on Sullivan et al. 1990) was used in the last sea scallop benchmark assessment in 2007 as the primary methodology. A slightly refined version of this model is used in this assessment as well (Table B-1).

Life History and Distribution

Sea scallops are found in the Northwest Atlantic Ocean from North Carolina to Newfoundland along the continental shelf, typically on sand and gravel bottoms (Hart and Chute 2004). Sea scallops feed by filtering phytoplankton, microzooplankton, and detritus particles. Sexes are separate and fertilization is external. Sea scallops typically become mature at age 2, but gamete production is limited until age 4. Larvae are planktonic for 4-7 weeks before settling to the bottom. Scallops fully recruit to the NEFSC survey at 40 mm SH, and to the current commercial fishery at around 90-105 mm SH, although sea scallops between 70-90 mm were common in landings prior to 2000.[3]

According to Amendment 10 of the Atlantic Sea Scallop Fishery Management Plan, all sea scallops in the US EEZ belong to a single stock. However, the US sea scallop stock can be divided into Georges Bank, Mid-Atlantic, Southern New England, and Gulf of Maine regional components based on survey data, fishery patterns, and other information (NEFSC 2004, Figure B-1). For assessment modeling purposes, Southern New England is considered to be part of the Georges Bank region.

Age and growth

Sea scallop assessments prior to 2007 estimated growth using the von Bertalanffy growth parameters from Serchuk et al. (1979). During the 2007 assessment, new analysis of shells collected during the 2001-2006 NEFSC scallop surveys was introduced (NEFSC 2007). This approach was based on growth increments inferred by successive rings on shells. The shell rings have been confirmed as annual marks (NEFSC 2007, Hart and Chute 2009a). Von Bertalanffy growth parameters were estimated in NEFSC (2007) using data from surveys from 2001 to 2006 using a mixed-effects model. Hart and Chute (2009b) gave a slightly refined version of this model that also included shells collected in 2007. Here we updated these estimates to include shells collected in 2008, using the same methodology as Hart and Chute (2009b). The current growth curves have lower mean L_∞ and higher mean K values than Serchuk et al. (1979). Differences between the current estimates and that of NEFSC (2007) are minor, and that between current estimates and Hart and Chute (2009b) are almost negligible (Figure B-2). Note that growth parameter t_0 cannot be estimated using growth increments, but it is not used in this assessment.

[3] Scallop body size is measured as shell height (SH, the maximum distance between the umbo and shell margin).

Mean growth parameters for sea scallops

Source	Region	L_∞	SE	K	SE
New (NEFSC 2010)	Mid-Atlantic	132.1	0.3	0.527	0.004
	Georges Bank	144.0	0.3	0.429	0.002
Hart & Chute (2009)	Mid-Atlantic	133.3	0.4	0.508	0.004
	Georges Bank	143.9	0.3	0.427	0.002
NEFSC (2007)	Mid-Atlantic	131.6	0.4	0.495	0.004
	Georges Bank	146.5	0.3	0.375	0.002
Serchuk et al. (1979)	Mid-Atlantic	151.84		0.2997	
	Georges Bank	152.46		0.3374	

Maturity and fecundity

Sexual maturity commences at age 2; sea scallops > 40 mm that are reliably detected in the surveys used in this assessment are all considered mature individuals. Although sea scallops reach sexual maturity at a relatively young age, individuals younger than 4 years may contribute little to total egg production (MacDonald and Thompson 1985; NEFSC 1993).

According to MacDonald and Thompson (1985) and McGarvey et al. (1992), annual fecundity (reproductive output, including maturity, spawning frequency, oocyte production, etc.) increases quickly with shell height in sea scallops

(*Eggs*=0.00000034 $SH^{4.07}$). Spawning generally occurs in late summer or early autumn. DuPaul et al. (1989) found evidence of spring, as well as autumn, spawning in the Mid-Atlantic Bight area. Almeida et al. (1994) and Dibacco et al. (1995) found evidence of limited winter-spring spawning on Georges Bank.

Shell height/meat weight relationships

Shell height-meat weight relationships allow conversion from numbers of scallops at a given size to equivalent meat weights. They are expressed in the form *W*=exp(• +• ln(*H*)), where *W* is meat weight in grams and *H* is shell height in mm. NEFSC (2001) combined the shell height/meat weight relationships from Serchuk and Rak (1983) with relationships from NEFSC (1999; later published as Lai and Helser 2004) to obtain “blended” estimates that were used in NEFSC (2001) and NEFSC (2004).

New shell height/meat weight data was collected during annual NEFSC sea scallop surveys during 2001-2009. Unlike previous studies, where meats were either frozen or brought in live and then weighed on land, meats were weighted at sea just after they were shucked. Estimates based on the 2001-2006 data were used in NEFSC (2007). This assessment updates these estimates by adding 2007-2008 data (see table below,

Figure B-3 and Appendix VII). Due to the change in timing of the survey, 2009 data were not used.

Meat weights also depend on covariates such as depth and latitude. Meat weights decreasing with depth, probably because of reduced food (phytoplankton) supply. Analysis of the new data indicated that depth and (at least in some cases) latitude had a significant effect on the the shell height/meat weight relationship (Appendix VII). Estimated coefficients for the relationship *W*=exp(• +[• +• ln(*D*)]ln(*H*) + γln(*D*)+• ln(*L*)), where *D* is depth in meters and *L* is latitude, are given below. In this assessment, depth-adjusted shell height/meat weight relationships were used to calculated survey biomass information, and traditional relationships were used in the models (CASA and SAMS), where depth is not explicit.

	α	β	γ	δ	ρ
Mid-Atlantic Bight					
Haynes (1966)	-11.09	3.04			
Serchuk and Rak (1983)	-12.16	3.25			
NEFSC (2001)	-12.25	3.26			
Lai and Helser (2004)	-12.34	3.28			
NEFSC (2007)	-12.01	3.22			
NEFSC (2007) with Depth effect	-9.18	3.18	-0.65		
NEFSC (2010)	-10.80	2.97			
NEFSC (2010) with Depth effect	-8.94	2.94	-0.43		
NEFSC (2010) with Depth effect and interaction	-16.88	4.64	1.57	-	-0.43
Georges Bank					
Haynes (1966)	-10.84	2.95			
Serchuk and Rak (1983)	-11.77	3.17			
NEFSC (2001)	-11.60	3.12			
Lai and Helser (2004)	-11.44	3.07			
NEFSC (2007)	-10.70	2.94			
NEFSC (2007) with Depth effect	-8.62	2.95	-0.51		
NEFSC (2010)	-10.25	2.85			
NEFSC (2010) with Depth effect	-8.05	2.84	-0.51		
NEFSC (2010) with Depth, Latitude and subarea effect	14.380	2.826	-0.529	5.980	0.051[b]

Meat weights for scallops in the commercial fishery may differ from those predicted based on research survey data for a number of reasons. First, the shell height-meat weight relationship varies seasonally, in part due to the reproductive cycle, so that meat weights collected during the NEFSC survey in July and August may differ from those in the rest of year. Additionally, commercial fishers concentrate on speed, and often leave some meat on the shell during shucking (Naidu 1987, Kirkley and DuPaul 1989). On the other hand, meats may gain weight due to water uptake during storage on ice (DuPaul et al. 1990). Finally, fishers may target areas with relatively large meat weight at shell height, and thus may increase commercial meat weights compared to that collected on the research vessel.

Observer data was used to adjust meat weights for seasonal variation and for commercial practices (Appendix VIII). Annual commercial meat weight anomalies were computed based on the seasonal patterns of landings together with the mean monthly commercial meat weight at shell height (Figure B-4).

Natural mortality

Previous assessments assumed a natural mortality of M = 0.1 based on Merrill and Posgay (1964), who estimated M based on ratios of clappers to live scallops in survey data. Clappers are shells from dead scallops that are still intact (i.e., both halves still connected by the hinge ligament). The basis of the estimate (Dickie 1955) is an assumed balance between the rate at which new clappers are produced ($M \cdot L$, where L is the number of live scallops) and the rate at which clappers separate ($S \cdot C$, where S is the rate at which shell ligaments degrade, and C is the number of clappers). At equilibrium, the rates of production and loss must be equal, so that $M \cdot L = S \cdot C$ and:

$M=C/(L \cdot S).$

Merrill and Posgay estimated S=33 weeks from the amount of fouling on the interior of clappers. The observed ratio C/L was about 0.066 and M was thus estimated to be 0.104 • 0.1 y^{-1}. However, the estimate of S is highly uncertain; for example Dickie (1955) estimated S to be 14.3 weeks based on tank experiments. The high level of uncertainty in the denominator implies that the estimator for M using the point estimated of S is biased low. If the standard error in the estimate of S is 12 weeks, an unbiased estimate of M is slightly more than 0.12. For this assessment, we use an estimate of M = 0.12 for Georges Bank. As shown below, this new assumption is supported by a number of modeling results.

No direct estimate of M is available for Mid-Atlantic sea scallops. The ratio of the growth coefficient K to M is generally regarded as a life history invariant that should be approximately constant for similar organisms (Beverton and Holt 1959, Chernov 1993). Applying this idea indicates that sea scallop natural mortality in the Mid-Atlantic should be about 0.527/0.429 that of Georges Bank (see the estimates of growth coefficients above). Using M = 0.12 in Georges Bank implies that natural mortality in the Mid-Atlantic is 0.12*0.527/0.429, or about 0.15. This is the estimate used in this assessment.

TOR 1: Commercial and Recreational Catch

The US sea scallop fishery is currently conducted mainly by about 350 vessels with limited access permits. Two types of allocation are given to each vessel. The first are trips (with a trip limit, typically of 18,000 lbs meats) to rotational access areas that had been closed to scallop fishing in the past. The second are days at sea, which can be used in areas outside the closed and access areas. Vessels fishing under days at sea are restricted to a 7 man crew in order to limit their processing power. The percentage of landings from the access trips have increased since the access area programs began in 1999; in recent years, about 60% of landings are from the access areas. Landings from 1964-2009 are given in Table B-2.

The remainder of landings come from vessels operating under "General Category" permits that are restricted to 400 lbs per trip, with a maximum of one trip per day. Landings from these vessels were less than 1% of total landings in the late 1990s, but increased to 10% or more of landings during 2007-2009. This type of permit had been open access, but was converted to an individual transferable quota (ITQ) fishery in March 2010.

Principal ports in the sea scallop fishery are New Bedford, MA, Cape May, NJ, and Hampton Roads, VA. New Bedford style scallop dredges are the main gear type in all regions, although some scallop vessels use otter trawls in the Mid-Atlantic. Recreational catch is negligible; a small amount of catch in the Gulf of Maine may be due to recreational divers.

Management history

The sea scallop fishery in the US EEZ is managed under the Atlantic Sea Scallop Fishery Management Plan (FMP), implemented on May 15, 1982. From 1982 to 1994, the primary management control was a minimum average meat weight requirement for landings.

FMP Amendment 4 (NEFMC 1993), implemented in 1994, changed the management strategy from meat count regulation to limited access, effort control and gear regulations for the entire US EEZ. Incremental restrictions were made on days-at-sea (DAS), minimum ring size, and crew limits (Table B-3). In addition, three large areas on Georges Bank and Nantucket Shoals were closed to groundfish and scallop fishing in December 1994 (Figure B-1). Scallop biomass rapidly increased in these areas. Two areas in the Mid-Atlantic were closed to scallop fishing in April 1998 for three years in order to similarly increase scallop biomass and mean weight.

Sea scallops were formally declared overfished in 1997, and Amendment 7 was implemented during 1998 with more stringent days-at-sea limitations and a mortality schedule intended to rebuild the stocks within ten years. Subsequent analyses considering effects of closed areas indicated that the stocks would rebuild with less severe effort reductions than called for in Amendment 7, and this days at sea schedule was thus modified. A combination of the closures, effort reduction, gear and crew restrictions led to a rapid increase in biomass (Hart and Rago 2006), and sea scallops were rebuilt by 2001. Prior to 2004, there were a number of ad hoc area management measures, including the Georges Bank and Mid-Atlantic closures in 1994 and 1998, limited reopenings of portions of the Georges Bank areas between June 1999 and January 2001, and reopening of the first Mid-Atlantic rotational areas in 2001.

A new set of regulations was implemented as Amendment 10 during 2004. This amendment formalized an area based management system, with provisions and criteria for new rotational closures, and separate allocations (in days-at-sea or TACs) for reopened closed areas and general open areas. Amendment 10 closed an area offshore of Delaware Bay (the Elephant Trunk area) where high numbers of small scallops were observed in the 2002 and 2003 surveys. This area reopened in 2007, when an area directly to the south was closed (Delmarva closure). One of the original Mid-Atlantic rotational closures, Hudson Canyon South, which had been closed in 1998 and reopened in 2001, was closed again in 2008, and is scheduled to reopen in 2011.

Amendment 10 also increased the minimum ring size to 4” and, together with subsequent frameworks, allowed limited reopening of portions of the groundfish closed areas.

Landings

Sea scallop landings in the US increased substantially after the mid-1940’s (Figure B-5), with peaks occurring around 1960, 1978, 1990, and 2004. Maximum US landings were 29,109 mt meats in 2004.

Proration of total commercial sea scallop landings into Georges Bank, Mid-Atlantic, Southern New England, and the Gulf of Maine regions used the standard allocation procedures of the NEFSC (Wigley et al. 2008). Landings from the Georges Bank and the Mid-Atlantic regions have dominated the fishery since 1964 (Table B-2 and Figure B-6). US Georges Bank landings had peaks during the early 1960’s, around 1980 and 1990, but declined precipitously during 1993 and remained low through 1998 (Table B-2 and Figure B-6). Landings in Georges Bank during 1999-2004 were fairly steady, averaging almost 5000 mt annually, and then increased in 2005-2006, primarily due to reopening of portions of the groundfish closed areas to scallop fishing. Poor recruitment in the middle of the decade and the reduction of biomass in the Georges Bank access areas have led to reductions in landings in the most recent years.

Until recently, the Mid-Atlantic landings were lower than those on Georges Bank. Mid-Atlantic landings during 1962-1982 averaged less than 1800 mt per year. An upward trend in both recruitment and landings has been evident in the Mid-Atlantic since the mid-eighties. Landings peaked in 2004 at 24,494 mt.

Landings from other areas (Gulf of Maine and Southern New England) are minor in comparison (Table B-2). Most of the Gulf of Maine scallop population is assessed and managed by the State of Maine because it is primarily in state waters (see Appendices V and VI). Gulf of Maine landings in 2009 were less than 1% of the total US sea scallop landings. Maximum landings in the Gulf of Maine were 1,614 mt during 1980.

Fishing effort and LPUE

Prior to 1994, landings and effort data were collected during port interviews by port agents and based on dealer data. Since 1994, commercial data are available as dealer reports (DR) and in vessel trip report (VTR) logbooks. DR data are total landings, and, since 1998, landings by market category. VTR data contain information about area fished, fishing effort, and retained catches of sea scallops. Ability to link DR and VTR reports in data processing is reduced by incomplete data reports and other problems, although there have been significant improvements recently. A standardized method (Wigley et al. 2008) for matching DR to VTRs and assigning areas to landings was used to allocate landings to region for 1994-2008. The method used in previous assessments (e.g., NEFSC 2007) that stratified landings and VTR by state was used for 2009, since the allocation tables for 2009 have not yet been completed.

Landings per unit effort (LPUE, computed as landings per day fished) (Figure B-7) shows a general downward trend from the beginning of the time series to around 1998, with occasional spikes upward probably due to strong recruitment events. LPUE increased considerably from 1999-2003 as the stock recovered; further increases in LPUE have been seen in recent years in the Mid-Atlantic, likely due to strong recruitment. Note the close correspondence in most years between the LPUE in the Mid-Atlantic and Georges Bank, probably reflecting the mobility of the fleet; if one area has higher catch rates, it is fished harder until the rates are equalized. Although comparisons of LPUE before and after the change in data collection procedures during 1994 need to be made cautiously, there is no clear break in the LPUE trend in 1994.

Fishing effort (days fished) in the US sea scallop fishery generally increased from the mid-1960s to about 1991, and then decreased during the 1990s, first because of low catch rates, and later as a result of effort reduction measures (Figure B-8). Effort increased in the Mid-Atlantic during 2000-2005, initially due to reactivation of latent effort among limited access vessels, and then due to increases in general category effort. Total effort since 2005 has remained fairly stable, though there have been shifts between regions.

Discards and discard mortality

Sea scallops are sometimes discarded on directed scallop trips because they are too small to be economically profitable to shuck, or because of high-grading, particularly during access area trips. Ratios of discard to total catch (by weight) were recorded by sea samplers aboard commercial vessels since 1992, though sampling intensity on non-access area trips was low until 2003; see Appendix II for detailed estimates.

Discarded sea scallops may suffer mortality on deck due to crushing, high temperatures, or desiccation. There may also be mortality after they are thrown back into the water from physiological stress and shock, or from increased predation due to shock and inability to swim or shell damage (Veale et al. 2000, Jenkins and Brand 2001). Murawski and Serchuk (1989) estimated that about 90% of tagged scallops were still living several days after being tagged and placed back in the water. Total discard mortality (including mortality on deck) is uncertain but has been estimated as 20% in previous assessments (e.g., NEFSC 2007); this assessment also makes this assumption. However, discard mortality may be higher during the Mid-Atlantic during the summer due to high water and deck temperatures.

Incidental mortality

Scallop dredges likely kill and injure some scallops that are contacted but not caught, primarily due to damage (e.g., crushing) caused to the shells by the dredge. Caddy (1973) estimated that 15-20% of the scallops remaining in the track of a dredge were killed. Murawski and Serchuk (1989) estimated that less than 5% of the scallops remaining in the track of a dredge suffered non-landed mortality. Caddy's study was done in a relatively hard bottom area in Canada, while the Murawski and Serchuk study was in sandy bottom off the coast of New Jersey. It is possible that the difference in indirect mortality estimated in these two studies was due to different bottom types (Murawski and Serchuk 1989).

In order to use the above estimates to relate landed and non-landed fishing mortality in stock assessment calculations, it is necessary to know the efficiency e of the dredge (the probability that a fully recruited scallop in the path of a dredge is captured). Denote by c the fraction of scallops that suffer mortality among sea scallops in the path of the dredge but not caught. The best available information indicates that c = 0.15-0.2 (Caddy 1973), and $c < 0.05$ (Murawski and Serchuk 1989). The ratio R of scallops in the path of the dredge that were caught, to those killed but not caught is:

$$R = e/[c(1-e)]$$

If scallops suffer direct (i.e., landed) fishing mortality at rate F_L, then the rate of indirect (non-landed) fishing mortality will be (Hart 2003):

$$F_I = F_L / R = F_L\, c\, (1-e)/e.$$

If, for example, the commercial dredge efficiency e is 50%, then $F_I = F_L\, c$, where F_L is the fully recruited fishing mortality rate for sea scallops. Assuming c = 0.15 to 0.2 (Caddy 1973) gives $F_I = 0.15\, F_L$ to $0.2\, F_L$. With $c < 0.05$ (Murawski and Serchuk 1989) $F_I < 0.05\, F_L$. Because there may be unobserved damage, actual incidental mortality may be higher than that observed in these studies. For this assessment, incidental mortality was assumed to be $0.2\, F_L$ in Georges Bank and $0.1\, F_L$ in the Mid-Atlantic.

Commercial shell height data

Since most sea scallops are shucked at sea, it has often been difficult to obtain reliable commercial size compositions. Port samples of shells brought in by scallopers have been collected, but there are questions about whether the samples were representative of the landings and catch. Port samples taken during the meat count era often appear to be selected for their size rather than being randomly sampled, and the size composition of port samples from 1992-1994 differed considerably from those collected by at-sea observers during this same period. For this reason, size compositions from port samples after 1984 when meat count regulations were in force are not used in this assessment.

Sea samplers (observers) have collected shell heights of kept scallops from commercial vessels since 1992, and discarded scallops since 1994. Although these data are likely more reliable than that from port sampling, they still must be interpreted cautiously for years prior to 2003 (except for the access area fisheries) due to limited observer coverage.

Shell heights from port and sea sampling data indicate that sea scallops between 70-90 mm often made up a considerable portion of the landings during 1975-1998, but sizes selected by the fishery have increased since then, so that scallops less than 90 mm were rarely taken during 2002-2009 (Figure B-9).

Dealer data (landings) have been reported by market categories (under 10 meats per pound, 10-20 meats per pound, 20-30 meats per pound etc) since 1998 (Figure B-10). These data also indicate a trend towards larger sea scallops in landings. While nearly half the landings in 1998 were in the smaller market categories (more than 30 meats per pound), about 75% of the 2009 landings were below 20 count and about 99% were below 30 count.

Economic trends in the U.S. sea scallop fishery

This section describes the trends in landings, revenues, prices, producer surplus and profits for the sea scallop fishery since 1994.

Trends in landings, prices and revenues

In the fishing years 2002-2008, the landings from the northeast sea scallop fishery stayed above 50 million pounds, surpassing the levels observed historically (Figure B-11). The recovery of the scallop resource and consequent increase in landings and revenues was striking given that average scallop landings per year were below 16 million pounds during the 1994-1998 fishing years, less than one-third of the present level of landings. The increase in the abundance of scallops coupled with higher scallop prices increased the profitability of fishing for scallops by the general category vessels. As a result, general category landings increased from less than 0.4 million pounds during the 1994-1998 fishing years to more than 4 million pounds during the last four fishing years (2005-2008), peaking at 7 million pounds in 2005 or 13.5% of the total scallop landings.

Figure B-12 shows that total fleet revenues tripled from about $100 million in 1994 to over $350 million in 2008 (in inflation-adjusted 2008 dollars). Scallop ex-vessel prices increased after 2001 as the composition of landings changed to larger scallops that in general command a higher price than smaller scallops. However, the rise in prices was not the main factor that led to the increase in revenue in the recent years compared to 1994-1998 and in fact, the inflation adjusted ex-vessel price of scallops in 2008 was lower than the price in 1994 (Figure B-12). The increase in total fleet revenue was mainly due to the increase in scallop landings and the increase in the number of active limited access vessels during the same period. Fig B6-9 shows that average landings and revenue per limited access vessel more than doubled in recent years compared to the period 1994 -1998. The number of active limited access vessels increased by 50 % (from about 220 in 1994 to 345 in fishing year 2008) resulting in tripling of total fleet scallop landings and revenue in 2008 compared to 1994 (Figure B-12 and Figure B-13).

Figure B-13 shows that average scallop revenue per limited access vessel more than doubled from about $400,000 in 1994 to about $950,000 despite the fact that inflation adjusted ex-vessel price per pound of scallops was slightly higher in 1994 ($7.15 per pound) compared to the ex-vessel price in 2008 ($6.92 per pound). In other words, the doubling of revenue was the result of the doubling of the average scallop landings per vessel in 2008 (over 136,000 pounds) from its level in 1994 (over 57,000 pounds). The total fleet revenue for all the limited access vessels more than tripled during the same years as new vessels became active. Average scallop revenue per full-time vessel peaked in the 2005 fishing year to over $1.1 million as a result of higher landings combined with an increase in ex-vessel price to about $8.50 per pound of scallops (in terms inflation adjusted 2008 prices).

Trends in the meat count and size composition of scallops

Average scallop meat count has declined continuously since 1999 as a result of effort-reduction measures, area closures, and an increase in ring sizes implemented by the Sea Scallop FMP. The share of larger scallops increased with the share of U10 scallops rising to over 20% since 2006. The share of 11-20 count scallops increased from 12% in 1999 to 53% in 2008. On the other hand, the share of 30 or more count scallops declined from 30% in 1999 to 1% in 2008 (Figure B-10 and tables below). Larger scallops priced higher than the smaller scallops contributed to the increase in average scallop prices in recent years despite larger landings (Figure B-12 and tables below).

Size composition of scallops

YEAR	Under 10 count	11-20 count	21-30 count	30 count and over	Unclassified
1999	17%	12%	25%	35%	12%
2000	7%	18%	44%	20%	11%
2001	3%	24%	49%	11%	13%
2002	5%	15%	65%	5%	11%
2003	6%	21%	56%	3%	13%
2004	7%	41%	42%	2%	8%
2005	13%	57%	21%	2%	7%
2006	23%	52%	18%	1%	6%
2007	24%	52%	13%	4%	8%
2008	23%	53%	18%	1%	4%

Price of scallop by market category (in 2008 inflation adjusted prices)

YEAR	<=10 count	11-20 count	21-30 count	>30 count
1999	7.8	7.9	7.3	6.4
2000	8.7	6.8	5.9	6.1
2001	7.2	4.7	4.4	4.7
2002	6.7	4.8	4.5	5.1
2003	5.7	4.8	4.8	5.3
2004	6.8	5.8	5.5	5.7
2005	8.8	8.6	8.5	8.3
2006	6.6	7.3	7.6	7.6
2007	7.2	6.9	6.8	6.2
2008	7.2	6.9	6.8	6.4

Trends in Foreign Trade

One of most significant change in the trend for foreign trade for scallops after 1999 was the striking increase in scallop exports. The increase in landings especially of larger scallops led to a tripling of U.S. exports of scallops from about 5 million lb. in 1999 to over 20 million lb. per year since 2005 (Figure B-14). Figure B-14 shows exports from New England and Mid-Atlantic ports combined including fresh, frozen and processed scallops. Although exports include exports of bay, calico or weathervane scallops, it mainly consists of sea scallops. France and other European countries were the main importers of US scallops. The exports from all other states and areas totaled only about $1 million in 2006 and 2007, and thus were not considered significant. Imports of scallops fluctuated between 45 million lb. and 60 million lb. during the same period.

TOR 2: Survey Data

Sea scallop surveys were conducted by NEFSC in 1975 and annually after 1977 to measure abundance and size composition of sea scallops in the Georges Bank and Mid-Atlantic regions (Figure B-1). The 1975-1978 surveys used a 3.08 m (10') unlined dredge with 50 mm rings. A 2.44 m (8') survey dredge with 50 mm rings and a 38 mm plastic liner has been used consistently since 1979. The lined survey dredge was judged to be unselective for scallops greater than 40 mm by comparing its catches to observations from sea floor video (NEFSC 2007). The northern edge of Georges Bank was not surveyed until 1982, so survey data for this area are incomplete for this area during 1975-1981. The 1979-1981 data were supplemented with Canadian survey data that covered much of the unsurveyed area (see Appendix XIII), allowing an extension of the lined survey dredge time series back to 1979.

The *R/V Albatross IV* was used for all NEFSC scallop surveys from 1975-2007, except during 1990-1993, when the *R/V Oregon II* was used instead. Surveys by the *R/V Albatross IV* during 1989 and 1999 were incomplete on Georges Bank. In 1989, the *R/V Oregon II* and *R/V Chapman* were used to sample the South Channel and a section of the Southeast Part. Serchuk and Wigley (1989) found no significant differences in catch rates between the *R/V Albatross IV*, *R/V Oregon II* and *R/V Chapman.*

The *F/V Tradition* was used to complete the 1999 survey on Georges Bank. NEFSC (2001) found no statistically significant differences in catch rates between the *F/V Tradition and R/V Albatross IV* from 21 comparison stations after adjustments were made for tow path length. Therefore, as in previous assessments (e.g., NEFSC 2004), survey indices for the period 1990-93 based on data from the *R/V Oregon II* were used without adjustment, and survey dredge tows from the *F/V Tradition* in 1999 were used after adjusting for tow distance.

In 2008-2009, the NEFSC scallop survey was conducted on the *R/V Hugh Sharp*. Direct and indirect comparisons between the catches of these vessels showed no significant differences (Appendix IV). However, examination of tow path length from dredge sensor data indicates that the tow path of the dredge on the R/V Sharp is about 5% longer than the R/V Albatross. Thus, survey catches in 2008-9 were reduced by 5%. Rock excluder chains have been used on the NEFSC sea scallop survey dredge since 2004 in certain hard bottom strata to enhance safety at sea and increase reliability (NEFSC 2004). Based on pair tows with and without the excluders, the best overall estimate was that rock chains increased survey catches on hard grounds by a factor of 1.31 (cv = 0.196). To accommodate rock chain effects in hard bottom areas, survey data collected prior to 2004 from strata 49-52 were multiplied by 1.31 prior to calculating stratified random means for larger areas; variance calculations in these strata include a term to account for the uncertainty in the adjustment factor (NEFSC 2007).

Calculation of mean numbers of scallops per tow, mean meat weight per tow and variances in this assessment were standard calculations for stratified random surveys (Serchuk and Wigley 1989; Wigley and Serchuk 1996; Smith 1997) with some extensions described below.

Relatively high abundance of sea scallops in closed areas makes it necessary to post-stratify survey data by splitting NEFSC shellfish strata that cross open/closed area boundaries. After post-stratification, adjacent strata were grouped into regions corresponding to the various open and closed areas. Finally, in cases where the closed or open portion of an NEFSC survey stratum was very small, it was necessary to combine the small portion with an adjacent stratum to form a new slightly larger stratum (NEFSC 1999).

Survey abundance and biomass trends

Biomass and abundance trends for the Mid-Atlantic Bight and Georges Bank are presented in Table B-4 and Figure B-15 and Figure B-16. Variances for strata with zero means were assumed to be zero.

In the Mid-Atlantic Bight, abundance and biomass were at low levels during 1979-1997, and then increased rapidly during 1998-2003, due to area closures, reduced fishing mortality, changes in fishery selectivity, and strong recruitment. Biomass was relatively stable since 2003. In Georges Bank, biomass and abundance increased during 1995-2000 after implementation of closures and effort reduction measures. Abundance and biomass declined from 2004-2007 because poor recruitment and reopening of portions of the groundfish closed areas. Abundances, and to a lesser extent, biomasses, increased since 2007 due to strong recruitment. Survey shell height frequencies show a trend to larger shell heights in both regions in recent years (Figure B-17).

Video survey data collected by the School for Marine Sciences and Technology (SMAST), University of Massachusetts, Dartmouth between 2003-2009 (Table B-5, Table B-6 and Figure B-18). SMAST survey data are counts and shell height measurements from images that were recorded by two video cameras. The “large” camera was mounted 1.575 m above the bottom in the center of the sampling frame while the “small” camera was mounted 0.7 m above the bottom. Adjustments have been made in this assessment to the estimated observed area of a quadrat, which is the area viewed by the large camera and to the number of sea scallops actually counted (Appendix III).

The SMAST survey is based on a systematic sampling pattern with stations centered on a 5.6 x 5.6 km grid pattern (Stokesbury et al. 2004). Four quadrats (drops) are sampled at each station and one image taken with each camera is analyzed from each quadrat. The sampling frame and cameras are placed on the bottom at the center of the grid where video footage from the first quadrat is collected. The sampling frame is then raised until the sea floor is no longer visible and the ship is allowed to drift approximately 50 m in the current before the sampling frame is lowered and video footage from the second quadrat image is collected. The third and fourth images are collected in the same manner. All scallops with any portion of their shell lying within the sample area are counted. Measurements are taken from images projected on a digitizing tablet from all specimens where the umbo and shell margins are clearly visible. The precision of measurements must be considered in interpreting video shell height data. Based on Jacobson et al. (2010) and NEFSC (2004), video shell height measurements from the large camera have a standard deviation of 6.1 mm across a wide range of sea scallop shell heights.

Video survey data in this assessment are expressed as densities (number m^{-2}). Variances for estimated densities are approximated using the estimator for a simple random survey applied to station means. There was some variability in the areas covered during each year (Table B-5 and Table B-6).

Dredge efficiency calibration

During 2007-2009, approximately 140 NEFSC scallop survey tows were also sampled using the HabCam towed digital camera system (Appendices IX and X). Analysis of these tows indicates that the lined survey dredge has an efficiency of about 0.44 in sandy areas and 0.38 in survey strata with a substantial fraction of gravel/cobble/rock substrate (Appendix X). These estimates are reasonably consistent with previous efficiency estimates (Table B-7).

TOR 3: Fishing Mortality, Biomass, and Recruitment Estimates

A catch at size analysis (CASA, Sullivan et al 1990) was used as the primary assessment model. CASA models growth using a stochastic growth matrix, which can be estimated using shell growth increment data. A CASA model for sea scallops was presented for preliminary review in (NEFSC 2004) and was used as the primary assessment model in the last assessment (NEFSC 2007). Simulation testing generally indicated good model performance (NEFSC 2007). CASA models for both stocks were run between 1975-2009. Shell heights were modeled with 5mm shell height bins starting at 20mm, but only scallops larger than 40mm were used in tuning to the data. The final (plus) group were the bins that included L_{∞} ; this bin were given special plus group weights based on the mean observed weight in the NEFSC survey in that year for scallops in the plus group (Figure B-19). Transition matrices were derived directly from shell increment data, as in the last assessment. Population shell height/meat weight conversions were based on 2001-2008 research vessel derived parameters, and fishery meat weights were adjusted based on estimated seasonal anomalies and the seasonal distribution of landings in that year (see Appendix VIII). Commercial shell heights data was obtained from 1975-1984 from port samples, and from 1992-2009 from sea samples (observers). Asymptotic delta method variances calculated in CASA with AD-Model Builder software were used to compute variances and coefficients of variation (cvs).

CASA model for Georges Bank

The model time-series for this assessment was 1975-2009, compared to 1982-2009 in NEFSC (2007). Three surveys were used for both trends and shell heights: the NEFSC lined dredge survey (1979-2009), the SMAST large video camera survey (2003-2009) and the NEFSC unlined dredge survey (1975-1978). The selectivity of the lined dredge survey was assumed flat (NEFSC 2007), and the selectivity of the video and unlined dredge survey was fixed on the basis of experimental evidence (NEFSC 2007, Serchuk and Smolowitz 1980). Priors with a cv of 0.15 were assumed for the NEFSC dredge (assuming a mean dredge efficiency of 0.41, see Appendix X), and for the large camera video survey (assuming 100% detectability of fully selected scallops). The prior distributions were implemented using symmetrical beta distributions. Fishery selectivity periods were 1975-1995, 1996-1998, 1999-2000, 2001-2003, and 2003-2009. Domed (double logistic) selectivity was assumed for the 1996-1998 and 2001-2003 periods, when there was no fishing access in the closed areas, so that large scallops were not fully selected to the fishery. LPUE was not used as an index of abundance. Natural mortality was set at $M = 0.12$ and incidental fishing mortality at 0.2 times fully recruited fishing mortality.

Model predicted trends and shell heights generally fit observations well (Figure B-20 to Figure B-23). This is also reflected in the relatively high implied effective sample sizes for the shell height data (Figure B-24). Mean posterior estimated efficiency for the lined dredge was 0.464, slightly higher than the 0.41 efficiency prior (Figure B-25). The large camera posterior mean was 1.5, indicating that the model estimates were lower than the camera data.

Fishery selectivity was strongly domed during the period that the closed areas were unavailable to the fishery (Figure B-26). Otherwise, selectivity has shifted over time toward larger shell heights. Biomass and abundance generally declined from 1975-1994 and then increased rapidly and reaching a peak in 2005 (Table B-8, Figure B-27). Biomass then fell through 2008, but increased from 2008 to 2009. Biomass in 2009 was 62470 mt. Recruitment appears to be cyclic, with several years of strong recruitment followed by several years of weaker recruitment. Fully recruited fishing mortality increased from 1975 to a peak of over 1.7 in 1992 and then declined. Fully recruited fishing mortality in 2009 was 0.18. As a result of the changes in selectivity and fully recruited fishing mortality, survival to large shell heights has increased substantially in recent years (Figure B-28). During 1975-1995, 100mm scallops were nearly fully selected, and 80 mm scallops were about 80% selected (Figure B-29). By contrast, 100 mm scallops were only about 40% selected during 2004-2009, whereas 80 mm scallops were essentially not selected at all.

Model abundance and biomass estimates correspond well to the expanded estimates from the lined dredge survey, but in most years are modestly below the large camera survey (Figure B-30). Model estimates of fishing mortality are consistent with the Beverton-Holt (1956) length-based equilibrium estimator (Figure B-31). The model 80+mm exploitation index (numbers caught/population numbers > 80mm), is similar to an empirical estimate of the same quantity, estimated directly from fishery and lined dredge survey data, expanded using a dredge efficiency of 0.41 (Figure B-31).

CASA Model for Mid-Atlantic

The Mid-Atlantic CASA model uses the same three survey time series as in Georges Bank, plus the NEFSC winter bottom trawl survey, conducted between 1992-2007. This survey uses "flat net" trawl gear similar to that used by commercial flounder and scallopers and should fairly reliably catch scallops. Preliminary runs with domed selectivity for this survey could not obtain reliable estimates for the declining portion of the dome, so selectivity was modeled by a logisitic curve with estimated parameters. However, residuals and direct comparisons between dredges and trawls (Rudders et al. 2000) suggest the possibility that some doming exists. Priors and selectivity assumptions for the other three surveys was as in Georges Bank. Selectivity periods were 1975-1979, 1980-1997, 1998-2001, 2002-2004, 2005-2009. The first period was modeled as domed (double logistic) selectivity due to the predominance of small scallops in fishery length data, whereas all the other periods were assumed to have logistic selectivity.

The model trend fit the lined dredge survey well, but was contrary to the large camera survey, which decreased while the model trend generally increased during 2003-2009 (Figure B-32). Predicted shell heights usually fit the data well, except for incoming strong year classes, which tended to be overestimated in the surveys relative to the model (Figure B-33, Figure B-34, Figure B-35, Figure B-36). Mean posterior efficiency for the dredge was 0.68, somewhat higher than the 0.44 estimated by the paired dredge/habcam experiment. Mean posterior efficiency for the large camera was 1.41, again indicating the model estimated abundances were generally less than those from the camera (Figure B-37). One cause of this is the downward trend in the large camera survey, which tends to pull the model estimate lower.

Selectivity was strongly domed during 1975-1979; selectivity moved father to the right during subsequent periods so that in the 2005-2009 period, only the plus group was fully selected (Figure B-38). Model estimated abundance and biomass were relatively low during 1975-1998, and then rapidly increased from 1998-2003 and has been steady to slightly increasing since then (Table B-8; Figure B-39). Recruitment has been much greater since 1998 than before this year. Fully recruited fishing mortality was between 0.5 and 1.2 in most years between 1975-1996. Since then, fishing mortality has ranged between 0.35 and 0.87. However, the force of fishing mortality is much less than this on most scallops because of the selectivity patterns. This is illustrated by the dramatic increase in survival since 1998 (Figure B-40), and the reductions in fishing mortality on 80 and 100 mm SH scallops (Figure B-41).

Model abundance and biomass estimates generally agree well with those of the lined dredge survey (expanded using a dredge efficiency of 0.44) except in the most recent period, when the dredge survey is modestly higher (Figure B-42). Model estimates were well below the large camera survey for 2003-2005, but well above them for 2009, again reflecting the conflicting trend. Model estimates of fishing mortality and exploitation agree reasonably well with simple empirical based estimates of these quantities, especially in the most recent years (Figure B-43).

Whole stock biomass, abundance and mortality

Biomass, egg production, abundance, recruitment and fishable mean abundance were estimated for the whole stock by adding estimates for the Mid-Atlantic Bight and Georges Bank. Whole stock fishing mortality rates for each year were calculated $F = \left(C_M + C_G\right) / \left(\overline{N}_M + \overline{N}_G\right)$ where C_M and C_V are catch numbers for the Mid-Atlantic Bight and Georges Bank. Terms in the denominator are average fishable abundances during each year calculated in the original CASA model $\overline{N} = \sum_L \frac{N_L\left(1 - e^{-Z_L}\right)}{Z_L}$ with the mortality rate for each size group (L) adjusted for fishery selectivity. The simple ratio formula used to calculate whole stock F is an "exact" solution because the catch equation implies that $C = F\overline{N}$.

Whole stock variances and coefficients of variation were calculated assuming that estimation errors for Georges Bank and the Mid-Atlantic Bight were independent. In particular, variances for biomass, abundance and catch estimates were the sum of the variances for Georges Bank and the Mid-Atlantic Bight. CVs for the ratios estimating whole stock F were approximated $CV_F = \sqrt{CV_C^2 + CV_{\overline{N}}^2}$, which is exact if catch number C_N and average abundance

$\overline{N}$ are independent and lognormally distributed (Deming 1960). The CV for measurement errors in catch for each region was 0.05, the same as assumed in fitting the CASA model.

Like the individual populations, whole-stock fishing mortality generally increased from 1975-1992 and then declined (Table B-8 and Figure B-44). Whole stock biomass, abundance and fishing mortality in 2009 were respectively 129,703 mt meats, 7446 billion (both on July 1) and 0.38. The biomass and abundance in 2009 were the highest in the 1975-2009 time series.

Variances for the stock as a whole depend on the assumption that model errors in Georges Bank and the Mid-Atlantic are independent; these variance would be higher if a positive correlation between model errors exists, and lower if they are negatively correlated.

The apparent precision of the estimates for sea scallops may be surprising and the cvs calculated in this assessment certainly do not capture all of the underlying uncertainties. Estimates were relatively precise because of the long time series of relatively precise dredge survey data and recent video survey data, together with the assumptions of known survey selectivities and prior information on survey efficiencies probably contributed to the small cvs. Retrospective and sensitivity analyses as well as likelihood profiles can help elucidate the uncertainties in the assessment.

Retrospective patterns

CASA model runs for Georges Bank and the Mid-Atlantic show moderate retrospective patterns, with biomass tending to decrease and fishing mortality tending to increase, with the additional years of data (Figure B-45 and Figure B-46). The pattern is stronger in the Mid-Atlantic, likely because of the downward re-estimation of the large year class observed in 2003 and the steep drop in the large camera survey in 2009.

Historical retrospective

Comparisons between the current estimates of fishing mortality and biomass and ones made in previous assessments indicate that estimates on Georges Bank have been fairly stable but there is a tendency in the Mid-Atlantic for estimates fishing mortality to increase and biomass to decrease over time (Figure B-47 and Figure B-48).

Likelihood profile analysis

Likelihood profiles were constructed for natural mortality (M) and mean of large camera survey q (Figure B-49 and Figure B-50). On Georges Bank, minimum –log-likehoods for natural mortality occur at about the estimated M = 0.12 for survey length compositions, and only slightly higher for survey trends, whereas the priors and commercial catches suggest a higher natural mortality. Most data sources tend to suggest a higher than estimated prior for the large camera survey.

In the Mid-Atlantic, survey trends and shell heights suggest the best estimate of natural mortality slightly below the estimated value (0.15), but the priors and commercial landings show minimums at larger values of M. Most sources of data tend to suggest a higher mean value for dredge efficiency than assumed in the prior, again demonstrating the tension between the survey priors and the other data sources.

Sensitivity analysis

The fact that survey estimated abundances tend to be somewhat higher than model estimates, especially in the Mid-Atlantic, suggest the possibility that there is some source of mortality, such as unreported landings, discard or incidental fishing mortality or natural mortality, the is greater than that assumed in the model. Alternatively, growth curves are based on data from the most recent period only (2001-2008); there would be model misspecification if growth was different in previous periods (e.g., because the heavy fishing affected growth, see the discussion in Hart and Chute 2009b). Violation of the assumption of spatial uniformity may also play a role in the conflict. Finally, it is possible that some systematic error in camera surveys could also explain at least part of the conflict (e.g., see Appendix III).

To estimate the uncertainty surrounding two key model imputs, sensitivity analyses were performed on input natural mortality and the assumed mean prior efficiencies of the lined dredge and large camera surveys (Figure B-51 and Figure B-52). For natural mortality, runs were conducting using the 5th, 25th, 75th, and 95th percentiles of the natural mortality distribution used in the stochastic reference point models.

Changing natural mortality modestly altered estimates, especially during the 1995-2005 period, but had little effect on the estimates of 2009 biomasses or fishing mortalities. Relaxing the assumptions on priors had almost no effect on 1975-1999 estimates, but did affect estimates in the most recent years, largely because that is when the large camera data occurs. Relaxing the priors gave lower biomasses and higher fishing mortalities than the basecase.

TOR 4: Biological Reference Points

In previous assessments, per recruit reference points F_{MAX} and B_{MAX} were used as proxies for F_{MSY} and B_{MSY}. F_{MAX} is the fishing mortality rate for fully recruited scallops that generates maximum yield-per-recruit. B_{MAX} was defined as the product of BPR_{MAX} (biomass per recruit at $F = F_{\mathrm{MAX}}$, from yield-per-recruit analysis) and median numbers of recruits. NEFSC (2007) reported January 1 biomass units, and estimated $F_{\mathrm{MAX}} = 0.29$ and $B_{\mathrm{MAX}} = 109{,}000$ mt meats as overall reference points, estimated from the CASA model.

Using the same methods but with updated data and CASA model, the estimates are $F_{\mathrm{MAX}} = 0.30$ and $B_{\mathrm{MAX}} = 127{,}000$ mt (Figure B-53). The increase in B_{MAX} is mostly due to the inclusion of special weights for the plus groups in the model; this feature was not in the 2007 model. The value of B_{MAX} is based on January 1 biomass, which was used to report biomass in NEFSC (2007). This assessment mainly reports model biomasses on July 1, which are less than those on January 1, because all growth and recruitment occur on that date in the model. The B_{MAX} corresponding to July 1 biomass is 85,000 mt. This value is somewhat less than the sum of the biomasses that maximize surplus production curves (Figure B-54).

As selectivity has shifted to larger scallops, yield per recruit curves have become increasingly flat, particularly in the Mid-Atlantic, making yield per recruit reference points both difficult to estimate and sensitive to small changes in parameters. Additionally, recruitment has been much stronger during the most recent period in the Mid-Atlantic when biomass has been high, suggesting that spawner-recruit relationships should be included in reference points.

This assessment introduces a stochastic model (SYM – Stochastic Yield Model) for calculating reference points and their uncertainty. It uses Monte-Carlo simulations to propagate the uncertainty of inputs to per recruit and stock-recruit calculations to the estimation of yield per recruit and yield curves. Besides its use in calculating limit reference points, a version of this model was employed to perform a risk assessment that was used to estimate Allowable Biological Catch (ABC) for the sea scallop fishery in 2010.

Description of stochastic yield model

Although the SYM model is separate from CASA, efforts were made to make the two models as compatible as possible. Recruits are initially spread out over 10 size bins (20-70 mm), and growth is modeled using a stochastic growth matrix, as in the CASA model.

Per recruit calculations depend on a number of parameters that each carry a level of uncertainty:

(1) Von Bertalanffy growth parameters K and L_∞

(2) Shell height/meat weight parameters a and b

(3) Natural mortality rate M

(4) Fishery selectivity parameters α and β

(5) The cull size of the catch and the fraction of discards that survive

(6) The level of incidental fishing mortality, i.e., non-catch mortality caused by fishing.

The mean, standard error and correlation (when applicable) for each of the parameters is given in Table B-9. Details on each of these parameters is given below.

Growth parameters K and L_∞ .

These were simulated as negatively correlated normals, using the mean and covariance from shell growth increment data, as estimated by a linear mixed-effects model (Hart and Chute 2009b), updated by including 2008 data. The level of individual variability in these two parameters was taken as estimated in the mixed-effects model without error.

Shell height/meat weight relationships.

Meat weight W at shell height H is calculated using a formula of the form:

$$W = \exp(a + b\ln(H)) \quad (1)$$

The means, variances and covariance of parameters a and b were taken from the analysis described in Appendix VII. Similar to the growth parameters, the estimates of a and b ahave a strong negative correlation. This means that the predicted meat weight at a given shell height carries less uncertainty than it would appear from the variances of the individual parameters. Meat weights vary seasonally, with the greatest meat weights during the late spring and early summer (NEFSC 2007). Haynes (1966) constructed a number of monthly shell height/meat weight relationships, and did not find any significant trend in the slopes. If this is the case, seasonality would not affect the F_{MAX} or F_{MSY} reference point. For this reason, seasonal variability was not considered a source of uncertainty for this analysis.

Natural mortality M.

As discussed in Section B3, natural mortality for sea scallops was estimated by Merrill and Posgay (1964) as

$$M = \frac{1}{S}\frac{C}{L} \quad (2)$$

where L is the number of live scallops, S is the mean clapper separation time and C is the number of clappers. Probably the greatest uncertainty in this calculation is the mean separation time S . For example, Dickie (1955) estimated S to be 100 days (14.3 weeks), less than half that estimated by Merrill and Posgay. Reflecting this uncertainty, it was assumed S was distributed as a gamma random variable, with mean 33 weeks and standard deviation 12 weeks. The resulting distribution of M has the desirable characteristic of being skewed to the right (Figure B-55). This makes sense since, for example, a natural mortality of $M = 0.2$ is possible, but an $M = 0$, or even close to zero, is not. Note that because S appears in the denominator of (2), the expected value of M is not equal to applying equation (2) with the mean value of S.

Fishery selectivity.

Fishery selectivity s was estimated using an ascending logistic curve of the form:

$$s = \frac{1}{1 + \exp(\alpha - \beta H)} \tag{3}$$

where H is shell height. The means and covariances of the α and β parameters were taken as estimated by the CASA stock assessment model during the most recent selectivity period. Note that fishery selectivity reflects targeting as well as gear selectivity.

Discard mortality .

Sea scallops that are caught but are less than 90 mm are assumed to be discarded, based on observer data. Sea scallops likely tolerate discarding fairly well, provided they are returned to the water relatively promptly and they are not damaged by the capture process or their time on deck. Here, discard mortality was simulated as a gamma distribution, with a mean of 0.2 and a standard deviation of 0.15, reflecting the high uncertainty in this parameter. This feature is not included in the CASA model, but makes little difference as few scallops below 90 mm are selected in the most recent selectivity period.

Incidental fishing mortality

Incidental fishing mortality occurs when scallops are killed but not captured by the gear. Consistent with the assumptions of the CASA model, incidental mortality was estimated as 0.2 that of landed fishing mortality on Georges Bank and 0.1 in the Mid-Atlantic. Because of the considerable uncertainty in these numbers, incidental mortality was simulated here with a gamma distribution with these means and coefficients of variation of 0.75.

Stock-recruit relationships

Stock-recruit relationships were based on the basecase CASA runs and fitted to Beverton-Holt stock-recruit curves of the form:

$$R = \frac{sB}{\gamma + B}, \tag{4}$$

assuming log-normal errors (Figure B-56). Here R is recruitment, B is spawning stock biomass (or egg production), and s and γ are parameters, representing the asymptotic recruitment when B is large, and the spawning stock biomass where recruitment is half its asymptotic value, respectively. Standard errors of the stock-recruit parameters and their correlation were also estimated using the delta method.

Calculation of equilibrium yield per recruit and yield

Per recruit and stock-recruit parameters were assigned probability distributions reflecting their level of uncertainty, as discussed above. For each iteration, parameters were drawn from their distributions, and then per recruit and yield curves were calculated. This was repeated for

$n = 50000$ iterations and the results collected. The stock-recruit parameters were simulated as correlated log-normals

For each run, equilibrium recruitment at fishing mortality F is given by

$$R = s - \gamma / b(F) \tag{5}$$

where b is biomass per recruit. Total yield is therefore

$$Y(F) = y(F)R = y(F)[(s-\gamma)/b(F)] \tag{6}$$

where y is yield per recruit.

Median (and mean) per recruit and yield curves were calculated as the median (mean) of these quantities as a function of fishing mortality. The probabilistic F_{MSY} (and F_{MAX} were taken as the fishing mortality that maximizes the median yield curve. The median was preferred because it avoided strong influence by likely unrealistic model outliers. The probabilistic MSY and B_{MSY} are the median yield and biomass at F_{MSY} over all runs.

Results

Simulated yield per recruit curves on Georges Bank generally showed a distinct peak between 0.2 and 0.3, but the simulated stock-recruit curves were almost completely flat (Figure B-57). By contrast, simulated yield per recruit curves from the Mid-Atlantic were flat, with F_{MAX} highly variable among runs, which induced a high F_{MAX} (0.835) for the median yield curve (Figure B-58). The correlation between biomass and recruitment induced a much lower F_{MSY} estimate (0.43) for the median yield curve for the Mid-Atlantic. The SYM model gives overall estimates of $F_{MSY} = 0.38$, $B_{MSY} = 125{,}358$ mt and MSY = 24,975 mt (Table B-9, Figure B-59).

Estimation of Allowable Biological Catch (ABC)

Probabilistic methods such as those employed here are ideal for quantifying risk and precaution, such as that used for deriving ABCs. For the purposes of setting the 2010 sea scallop ABC, the fishing mortality corresponding to the ABC was set by the NEFMC Science and Statistical Committee at the 25th percentile of the distribution of the overall F_{MSY} (i.e., the 25th of the distribution of F_{MSY} values from the individual simulations) which at the time was estimated at 0.28. Using the current simulations, the 25th percentile of F_{MSY} is at 0.31 (Figure B-59 (b)). Equilibrium yield at 0.31 is about 0.8% less than that at F_{MSY} (Figure B-60).

Special considerations for sedentary resources under area management

The above reference point calculations are based on the assumption that fishing mortality risk does not vary among individuals. For sedentary organisms such as sea scallops, these assumptions are never even approximately true; area management such as closed areas means that the assumption of uniform fishing mortality is strongly violated (Hart 2001, 2003; Smith and Rago 2004). In such situations, mean yield-per-recruit, averaged over all recruits, may be different than yield-per-recruit obtained by a conventional per-recruit calculation performed on a recruit that suffers the mean fishing mortality risk (Hart 2001). This condition is exaggerated, as in the case of the scallop fishery, with use of rotational or long-term closures. Moreover, estimates of fishing mortality may be biased low, because individuals with low mortality risk are overrepresented in the population (Hart 2001, 2003).

TOR 5: Status Determination

According to the Amendment 10 overfishing definition (NEFMC 2003), sea scallops are overfished when the survey biomass index for the whole stock falls below 1/2 B_{TARGET}. The target biomass estimated in NEFSC (2007) is B_{TARGET} = 109,000 mt (January 1) was calculated as the median recruitment in the survey time series times BPR_{MAX}, the biomass per recruit obtained when fishing at F_{MAX}. NEFSC (2007) estimated F_{MAX}= 0.29, which has been used since then as the overfishing threshold. The updated values are F_{MAX} = 0.30 and B_{MAX} = 85,000 mt (July 1 biomass). The new recommended stochastic MSY reference points are F_{MSY} = 0.38 and B_{MSY} = 125,358 mt.

According to the basecase CASA run, total biomass in 2009 was 129,703 mt meats, which is above the estimated B_{MSY} or its proxy, regardless of whether the previous, updated or proposed biomass target is used. Therefore, the sea scallop fishery was not overfished in 2009. The probability the stock was below the 1/2 B_{MSY} biomass threshold is < 0.0001, regardless of which biomass reference point is used.

Overall fishing mortality was 0.38 (to three decimal places 0.378), which is above the previous (NEFSC 2007) overfishing threshold of 0.29 and its updated value of 0.30, but equal to the newly recommended (in 2010) F_{MSY} = 0.38. Therefore, overfishing was not occurring in 2009 based on the new recommended overfishing definition; however, overfishing would be occurring if the previous definition or its updated value were to be used. Using the new recommended overfishing definition, the probability that overfishing was occurring in 2009 was just under 0.50.

TOR 6: Stock Projections

Because of the sedentary nature of sea scallops, fishing mortality can vary considerably in space even in the absence of area specific management (Hart 2001). Area management such as rotational and long-term closures can make variation even more extreme. Projections that ignore such variation might be unrealistic and misleading. For example, suppose 80% of the stock biomass is in areas closed to fishing (as occurred in some years in Georges Bank). A stock projection that ignored the closure and assumed a whole-stock F of 0.2 would forecast landings nearly equal to the entire stock biomass of the areas remaining open to fishing. Thus, using a non-spatial forecasting model can lead to setting a level of landings that appears sustainable if all areas were fished uniformly, but is in fact unsustainable for a given area management policy.

For this reason, a spatial forecasting model (the Scallop Area Management Simulator, SAMS) was developed for use in sea scallop management (Appendix XII). Various versions of SAMS have been used since 1999 and the model was discussed at length in the last assessment (NEFSC 2007). Growth is modeled in SAMS and CASA in a similar manner, except that each subarea of Georges Bank and the Mid-Atlantic in SAMS has its own stochastic growth transition matrix derived from the shell increments collected in that area. Mortality and recruitment are also area-specific. In example calculations, natural mortality was chosen from a gamma distribution with means 0.12 (Georges Bank) and 0.15 (Mid-Atlantic), to be compatible with reference point calculations in the SYM model (see Section B7). Fishing mortality can either be explicitly specified in each area, calculated using a simple fleet dynamics model which assumes fishing effort is proportional to fishable biomass, or a combination of the two.

Projected recruitment is modeled stochastically with the log-transformed mean and covariance for recruitment in each area matching that observed in NEFSC dredge survey time series. Initial conditions were based on the 2009 NEFSC and SMAST sea scallop surveys with uncertainty measured by bootstrapping as described by Smith (1997). Survey dredge efficiencies were set in SAMS so that the mean 2009 biomass matched estimates from the CASA model. Further details regarding the SAMS model are given in Appendix XII.

Example calculations

Only example calculations can be given here but the model has and will be used by the NEFMC Scallop Plan and Development Team to evaluate possible management alternatives, which are complex for sea scallops. For the example simulations, the stock area was split into 16 subareas (Figure B-61), six in the Mid-Atlantic (Virginia Beach, Delmarva, Elephant Trunk, Hudson Canyon South, New York Bight, and Long Island) and ten on Georges Bank (Closed Area I, II and Nantucket Lightship EFH closures, Closed Area I, II and Nantucket Lightship access areas, Great South Channel proposed closure and the remainder of the Great South Channel, Northern Edge and Peak, and Southeast Part).

The EFH (Essential Fish Habitat) closures on Georges Bank were assumed to be closed for the duration of the simulations. One of the Georges Bank access areas were assumed to be fished on a rotating basis (Closed Area II in 2009 and 2012, Nantucket Lightship in 2010 and 2013, and Closed Area I in 2011 and 2014). Landings in these areas (as actually has occurred or is planned) were set at 1400 mt in 2009, and 2700 mt in 2010-2014. The Hudson Canyon South rotational closure area was assumed to be closed to fishing in 2009-2010, and then reopened with a TAC of 5400 mt in 2011-2013. It is assumed to revert to a general open area in 2014. The Elephant Trunk rotational area was assumed to have landings of 8100 mt in 2009, 5400 mt in 2010 and 2700 mt in 2011, and then reverts to be part of the open areas. Landings in the Delmarva rotational area are assumed to be 2700 mt in 2009 and 2010, 5400 mt in 2011 and 2012 and then it reverts to the open pool. All other areas (Virginia Beach, New York Bight, Long Island, South Channel areas, Northern Edge and Peak, Southeast Part). In projections, fishing effort was allocated to areas so that the overall fishing mortality rate was 0.24 in all years, consistent with current policy, and somewhat lower than the 2009 recommend ABC fishing mortality of 0.28. Fishing effort was distributed among the open areas according to a simple fleet dynamics model, where fishing mortality in each area was assumed to be proportional to fishable biomass.

A total of n=5000 projection runs were performed, with stochastically varying initial conditions, recruitment, and natural mortality. Projected mean biomass is expected to increase modestly from 2009-2012, mainly on Georges Bank due to the large year classes observed during 2007-2009, and then level off (Figure B-62). Landings are expected to be lower in 2010 than 2009, then increase somewhat, with a peak in 2012 at about 27,000 mt, and then level off to about 24,000 mt. Fishing mortality is expected to be greater in the Mid-Atlantic than in Georges Bank. Not surprisingly, uncertainty regarding biomass and landings increases over time (Figure B-63). Nonetheless, the 25th percentile of biomass is over 130,000 mt in all years, and thus over the target biomass. The minimum biomass of the 5000 runs stayed above the overfishing threshold through 2012, but dropped below it for 2013 and 2014. However, even the 0.1th percentile of the runs remained over the overfishing threshold in all years. Thus, the forecasts indicate that there is little chance of the stock becoming overfished under status quo management.

In summary, the projections indicate that the stock is stable, and biomass and landings may increase modestly from 2009 levels assuming status quo management. Especially given the recent selectivity patterns, the stock's vulnerability to being overfished is low.

TOR 7: Research Recommendations

Research Recommendations from NEFSC 2007

1) Refine estimates of natural mortality focusing on variation among regions, size groups and over time. Abundance trends in closed areas where no fishing occurs may provide important information about the overall level of natural mortality and time trends. Survey clapper catches may provide information about spatial, temporal and size related patterns in natural mortality.
This assessment contains a re-evaluation of natural mortality in sea scallops. Further work on natural mortality using the closed areas is ongoing.
2) Evaluate the within and between reader error rates in identification and measurement of growth increments on scallop shells.
This has not been done since there is at this time only a single reader.
3) Improve estimates of incidental and discard mortality rates.
This has not been done, but the results of this assessment indicate its importance, especially for the Mid-Atlantic.
4) Consider using autocorrelated recruitment in SAMS projection model runs. CASA model estimates indicate that sea scallop recruitment may be autocorrelated.
SAMS has the ability to model autocorrelated recruitment, but this was not done in the simulations presented here because of the difficulties in estimating the autocorrelation on the small scale that SAMS operates.
5) Consider modeling the spatial dynamics of the fishing fleet in the SAMS projection model based on catch rates, rather than exploitable abundance, of scallops in each area.
Not done
6) Evaluate assumptions about the spatial dynamics of the fishing fleet in the SAMS model by comparing predicted distributions to VMS data.
Work with VMS data is ongoing, but has been slowed due to problems obtaining the data.
7) Investigate the feasibility and benefits of using information about the size composition of sea scallops in predicting the spatial distribution of the fishing fleet in the SAMS projection model.
Not done.
8) Evaluate the accuracy of the SAMS projection model retrospectively by comparison to historical survey abundance trends.
This has been done in other venues. The SAMS model had a tendency to overestimate projected biomass and landings. The changes in the assumptions of growth, natural mortality and incidental mortality may make the forecasts more realistic.
9) Consider implementing discard mortality calculations in the CASA model that are more detailed and involve discarded shell height composition data from at sea observers.
This was considered, but not done due to lack of time. Discard mortality may be important during some periods, especially in the Mid-Atlantic. Additionally, empirical studies estimating discard mortality will be needed to make the modeling useful.
10) Consider implementing a two or more "morph" formulation in the CASA model to accommodate scallops that grow at different rates.
Not done.

11) Consider approaches to implementing seasonal growth patterns in the CASA model to improve fit to shell height composition data. Scallops grow quickly at small sizes and growth rates vary by season.

Considerable time was spent on implementing a CASA model with seasonal growth, but the model did not perform well with seasonal growth. Thus, this assessment still uses an annual growth model.

New Research Recommendations

1. Look into a way to fit discarded scallops, which have a different length frequency from the rest of the population, into the model.
2. Evaluate the effect of the four-inch rings on incidental mortality. Now that a larger fraction of small scallops are traveling through the mesh, has incidental mortality increased or are the scallops relatively unscathed?
3. Consider finding a better way to express the variation in the HABCAM abundance data (the data were kriged for this assessment, and the variance was calculated by summing the variance of each of the krieged grids).
4. Look at the historical patterns of the “whole stock”; how the spatial patterns of scallops and the fishery have changed over time.
5. Estimate incidental mortality by running Habcam or an AUV along dredge tracks
6. Effort should be made to make sure the survey dredge is fitted with a camera at some point during the survey to record the movements of the dredge. This will help answer some questions about when the dredge starts and stops fishing, and the determination of tow times.
7. Seasonal patterns in scallop shell growth need to be analyzed and this data incorporated into the model.
8. Stock-recruit relationships should be calculated for various sub-sections of the stock, smaller areas than just MAB and GBK to look for possible patterns or relationships.
9. Further refine the estimate of the extent of scallop habitat relative to that of the survey
10. Age archived scallop shells from the 1980s and 1990s.
11. Continue to look at patterns of seasonality in weight of the meats and gonads, and timing of spawning.

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Tables

Table B-1. List of changes made to CASA models for the 2010 sea scallop assessment.

1) Updated growth increments -plus groups changed to match new *L*• estimates
2) Updated shell height meat weight relationships
3) Updated commercial meat weight anomalies (substantial changes)
4) Empirical plus group weights for fishery and population (new code and input data)
5) M=0.12 (GBK) and 0.15 (MAB) instead of 0.1
6) Incidental fishing mortality estimates increased (0.15 to 0.2 on GB; 0.04 to 0.1 in MAB)
7) NEFSC survey
   Adjustments for R/V Sharp in 2008-2009
   Canadian tows on GBK during 1979-1981
   Efficiency estimates from paired HabCam tows used as prior
   Used unlined dredge survey (1975+1977) on Georges Bank
8) GBK starts in 1975 (instead of 1982)
9) LPUE no longer used in model
10) SMAST large camera survey (2003-2009) in place of small camera
11) Prior cv(s) set at 0.15 rather than 0.20
12) Primarily report July 1 rather than January 1 abundance/biomass
13) Assumed CV for surveys tuned to residual variance

Table B-2. US sea scallop landings (mt meats) 1964-2009.

	Gulf of Maine				Georges Bank				S. New England				Mid Atlantic Bight				Total			
Year	dredge	trawl	other	sum	dredge	trawl	other	sum	dredge	trawl	other	sum	dredge	trawl	other	sum	dredge	trawl	other	sum
1964		0	208	208		0	6,241	6,241		52	3	55		0	137	137		52	6,590	6,642
1965		0	117	117		3	1,478	1,481		2	24	26		0	3,974	3,974		5	5,592	5,598
1966		0	102	102		0	883	884		0	8	8		0	4,061	4,061		1	5,055	5,056
1967		0	80	80		4	1,217	1,221		0	8	8		0	1,873	1,873		4	3,178	3,182
1968		0	113	113		0	993	994		0	56	56		0	2,437	2,437		0	3,599	3,599
1969		1	122	123		8	1,316	1,324		0	18	19		5	846	851		14	2,302	2,317
1970		0	132	132		5	1,410	1,415		0	6	6		14	459	473		19	2,006	2,026
1971		4	358	362		18	1,311	1,329		0	7	7		0	274	274		22	1,949	1,971
1972		1	524	525		5	816	821		0	2	2		5	653	658		11	1,995	2,006
1973		0	460	460		15	1,065	1,080		0	3	3		4	245	249		19	1,773	1,792
1974		0	223	223		15	911	926		0	4	5		0	937	938		16	2,076	2,091
1975		6	741	746		13	844	857		8	42	50		52	1,506	1,558		80	3,132	3,212
1976		3	364	366		38	1,723	1,761		4	3	7		819	2,972	3,791		361	5,061	5,422
1977		4	254	258		27	4,709	4,736		1	10	11		255	2,564	2,819		58	7,536	7,595
1978	242	1	0	243	5,532	37	0	5,569	25	2	0	27	4,435	207	0	4,642	10,234	247	0	10,481
1979	401	5	1	407	6,253	25	7	6,285	61	5	0	66	2,857	29	1	2,888	9,572	64	9	9,645
1980	1,489	122	3	1,614	5,382	34	2	5,419	130	3	0	133	2,202	85	79	2,366	9,204	245	83	9,532
1981	1,225	73	7	1,305	7,787	56	0	7,843	68	1	0	69	772	14	2	788	9,852	144	9	10,005
1982	631	28	5	664	6,204	119	0	6,322	126	0	0	126	1,602	6	2	1,610	8,562	153	7	8,723
1983	815	72	7	895	4,247	32	4	4,284	243	1	0	243	3,092	19	10	3,121	8,398	124	21	8,542
1984	651	18	10	678	3,011	29	3	3,043	161	3	0	164	3,695	53	2	3,750	7,518	103	14	7,635
1985	408	3	10	421	2,860	34	0	2,894	77	4	0	82	3,230	49	2	3,281	6,575	90	12	6,677
1986	308	2	6	316	4,428	10	0	4,438	76	2	0	78	3,407	386	6	3,799	8,218	400	12	8,631
1987	373	0	9	382	4,821	30	0	4,851	67	1	0	68	7,639	1,168	1	8,808	12,900	1,199	10	14,109
1988	506	7	13	526	6,036	18	0	6,054	65	4	0	68	6,071	938	8	7,017	12,678	966	21	13,666
1989	600	0	44	644	5,637	25	0	5,661	127	11	0	138	7,894	534	5	8,433	14,258	570	49	14,876
1990	545	0	28	574	9,972	10	0	9,982	110	6	0	116	6,364	541	10	6,915	16,991	558	38	17,587
1991	527	3	75	605	9,235	77	0	9,311	55	16	0	71	6,408	878	14	7,300	16,225	973	89	17,288
1992	676	2	45	722	8,230	7	0	8,238	119	5	0	124	4,562	570	5	5,137	13,587	584	50	14,221
1993	763	2	32	797	3,637	18	0	3,655	65	1	0	66	2,412	393	3	2,808	6,878	413	36	7,327
1994	410	6	9	425	1,182	7	0	1,189	29	1	0	30	5,211	754	0	5,965	6,832	768	9	7,609
1995	342	6	13	361	992	4	1	997	41	2	0	43	5,786	798	7	6,591	7,161	810	21	7,992
1996	544	5	12	561	2,126	7	4	2,137	59	5	0	64	4,467	653	4	5,124	7,196	670	20	7,886
1997	673	5	21	699	2,347	9	1	2,357	81	11	3	95	2,703	378	1	3,082	5,804	403	26	6,233
1998	392	5	15	412	2,045	19	1	2,065	103	3	0	106	2,411	564	6	2,981	4,951	591	22	5,564
1999	267	2	2	271	5,172	6	1	5,179	78	1	0	79	3,629	959	1	4,589	9,146	968	4	10,118
2000	162	21	43	226	4,910	40	5	4,955	85	3	1	89	8,139	1,210	2	9,351	13,296	1,274	51	14,621
2001	335	7	1	343	4,879	58	6	4,943	28	37	0	65	14,144	1,543	16	15,703	19,386	1,645	23	21,054
2002	386	18	1	405	5,967	33	11	6,011	20	12	0	32	15,981	1,426	36	17,443	22,354	1,489	48	23,891
2003	197	3	1	201	4,859	22	2	4,883	53	4	0	57	19,040	1,226	10	20,276	24,149	1,255	13	25,417
2004	165	12	0	177	4,249	146	11	4,406	830	151	11	992	22,313	1,194	26	23,533	27,557	1,503	48	29,108
2005	163	12	12	187	8,958	69	15	9,042	845	13	40	898	14,361	1,096	109	15,566	24,327	1,190	176	25,693
2006	147	3	5	155	15,688	51	21	15,760	2,029	10	8	2,047	7,944	782	46	8,772	25,808	846	80	26,734
2007	97	8	12	117	9,419	45	18	9,482	335	18	7	360	16,234	345	55	16,634	26,085	416	92	26,593
2008	103	12	5	120	6,405	24	11	6,440	303	6	16	325	16,819	556	13	17,388	23,630	598	45	24,273
2009	81	0	3	84	6,451	8	16	6,475	216	1	3	220	17,487	12	1,851	19,350	24,235	21	1,873	26,129

Table B-3. Summary of sea scallop management history.

Period	Days at sea#	Minimum Ring Size	Minimum Twine Top	Maximum Crew Size	GB Closures	GB Access Areas	MA Closures	MA Access Areas
1982-1993	N/A	N/A	N/A	N/A	0	0	0	0
1994	204	3"-3.25"	5.5"	9	3	0	0	0
1995	182	3.25"	5.5"	7	3	0	0	0
1996	182	3.5"	5.5"	7	3	0	0	0
1997	164	3.5"	5.5"	7	3	0	0	0
1998	142	3.5"	5.5"	7	3	0	2	0
1999	120	3.5"	5.5"	7	3	1	2	0
2000	120	3.5"	8"	7	3	3	2	0
2001	120	3.5"	8"	7	3	1	0	2
2002	120	3.5"	8"	7	3	0	0	2
2003	120	3.5"	8"	7	3	0	0	2
2004	42*	3.5"	8"	7	3	2	1	1
2005	40*	4"	10"	7	3	2	1	1
2006	52*	4"	10"	7	3	2	1	1
2007	51*	4"	10"	7	3	2	1	2
2008	35*	4"	10"	7	3	1	2	1
2009	37*	4"	10"	7	3	1	1	2

Full-time permit
*Does not include access area trips; for each year between 2005-2009, full-time vessels were allocated 5 access area trips, with trip limits of 18,000 lbs meats.

Table B-4. NEFSC sea scallop lined survey stratified mean indices for (a) Georges Bank, (b) Mid-Atlantic, and (c) combined for shell heights greater than 40 mm. The expanded abundance and biomass figures were calculated using an assumed efficiency of 0.41 for Georges Bank and 0.44 for the Mid-Atlantic.

(a) Georges Bank

year	Abundance index (mean N/tow)	CV	Biomass index (kg/tow)	CV	N tows	Proportion positive tows	mean weight (g/scallop)	Expanded abundance (millions)	Expanded biomass (mt)
1979	82.9	0.57	1.650	0.35	121	0.90	19.9	1042	20740
1980	70.2	0.32	0.785	0.16	155	0.78	11.2	883	9861
1981	46.4	0.20	0.957	0.18	86	0.80	20.6	583	12022
1982	133.3	0.56	0.837	0.31	129	0.80	6.3	1675	10517
1983	50.8	0.30	0.607	0.24	138	0.85	12.0	638	7626
1984	28.8	0.12	0.421	0.10	138	0.83	14.6	362	5294
1985	52.1	0.18	0.554	0.17	170	0.85	10.6	655	6967
1986	90.8	0.18	0.715	0.11	194	0.85	7.9	1141	8983
1987	107.0	0.21	0.907	0.17	190	0.82	8.5	1345	11402
1988	81.9	0.17	0.709	0.14	192	0.78	8.7	1029	8908
1989	85.0	0.35	0.702	0.16	254	0.82	8.3	1068	8818
1990	166.7	0.30	1.036	0.23	194	0.80	6.2	2095	13025
1991	242.2	0.49	1.116	0.26	194	0.88	4.6	3044	14031
1992	236.8	0.53	1.605	0.46	191	0.86	6.8	2976	20166
1993	57.5	0.29	0.400	0.17	182	0.82	7.0	722	5026
1994	38.4	0.18	0.367	0.13	194	0.80	9.6	482	4618
1995	109.2	0.25	0.649	0.17	193	0.85	5.9	1372	8159
1996	111.8	0.18	1.114	0.16	189	0.87	10.0	1406	14000
1997	78.7	0.14	1.292	0.15	206	0.85	16.4	989	16239
1998	265.6	0.26	3.728	0.33	230	0.86	14.0	3338	46850
1999	156.0	0.15	2.527	0.16	198	0.94	16.2	1961	31756
2000	681.2	0.30	6.118	0.21	188	0.89	9.0	8562	76893
2001	372.0	0.14	5.724	0.14	225	0.94	15.4	4676	71934
2002	294.8	0.15	6.158	0.14	229	0.90	20.9	3705	77398
2003	226.0	0.12	5.796	0.14	225	0.92	25.6	2840	72844
2004	264.2	0.11	7.606	0.13	230	0.92	28.8	3321	95596
2005	210.0	0.12	6.048	0.11	227	0.93	28.8	2640	76010
2006	153.5	0.11	5.013	0.14	237	0.91	32.6	1930	62999
2007	183.2	0.09	4.373	0.09	232	0.94	23.9	2303	54955
2008	292.9	0.13	6.242	0.10	182	0.90	21.3	3681	78448
2009	380.6	0.19	6.186	0.18	179	0.94	16.3	4784	77748

(b) Mid-Atlantic Bight

year	Abundance index (mean N/tow)	CV	Biomass index (kg/tow)	CV	N tows	Proportion positive tows	mean weight (g/scallop)	Expanded abundance (millions)	Expanded biomass (mt)
1979	32.3	0.09	0.580	0.10	166	0.92	17.9	466	8364
1980	41.2	0.12	0.497	0.08	167	0.94	12.1	595	7173
1981	30.7	0.16	0.386	0.12	167	0.91	12.6	443	5574
1982	31.2	0.11	0.406	0.08	185	0.91	13.0	451	5864
1983	29.1	0.09	0.365	0.08	193	0.89	12.5	420	5269
1984	29.4	0.10	0.351	0.08	204	0.91	12.0	424	5069
1985	69.9	0.12	0.558	0.08	201	0.94	8.0	1008	8048
1986	119.6	0.09	0.956	0.08	226	0.93	8.0	1726	13787
1987	119.9	0.09	0.829	0.06	226	0.93	6.9	1731	11962
1988	134.9	0.10	1.300	0.07	227	0.91	9.6	1946	18763
1989	171.1	0.09	1.190	0.07	244	0.93	7.0	2469	17175
1990	205.4	0.22	1.275	0.17	216	0.89	6.2	2964	18402
1991	77.0	0.10	0.738	0.11	228	0.92	9.6	1110	10647
1992	40.9	0.11	0.418	0.07	229	0.87	10.2	590	6037
1993	130.7	0.10	0.591	0.08	214	0.96	4.5	1886	8527
1994	128.0	0.11	0.787	0.09	227	0.94	6.1	1847	11351
1995	164.4	0.13	1.149	0.10	227	0.96	7.0	2372	16574
1996	55.8	0.08	0.568	0.07	211	0.89	10.2	806	8197
1997	42.5	0.13	0.423	0.06	225	0.93	10.0	613	6106
1998	151.8	0.17	0.841	0.14	227	0.92	5.5	2190	12132
1999	241.4	0.24	1.768	0.19	226	0.92	7.3	3483	25508
2000	294.1	0.15	3.060	0.13	229	0.88	10.4	4243	44156
2001	305.3	0.12	3.386	0.13	227	0.90	11.1	4405	48852
2002	295.0	0.10	3.721	0.11	206	0.89	12.6	4256	53694
2003	655.4	0.16	5.780	0.09	201	0.90	8.8	9456	83400
2004	494.5	0.12	5.332	0.07	248	0.89	10.8	7135	76938
2005	379.0	0.09	5.973	0.08	241	0.93	15.8	5469	86185
2006	380.1	0.09	5.926	0.07	230	0.93	15.6	5485	85505
2007	308.3	0.07	5.440	0.07	240	0.92	17.6	4449	78491
2008	435.5	0.10	6.229	0.09	207	0.96	14.3	6283	89884
2009	401.9	0.13	6.731	0.12	196	0.92	16.8	5798	97125

(c) Whole Stock									
year	Abundance index (mean N/tow)	CV	Biomass index (kg/tow)	CV	N tows	Proportion positive tows	mean weight (g/scallop)	Expanded abundance	Expanded biomass
1979	55.9	0.40	1.1	0.25	287	0.91	19.3	1508	29104
1980	54.7	0.20	0.6	0.10	322	0.86	11.5	1477	17033
1981	38.0	0.13	0.7	0.13	253	0.87	17.1	1026	17596
1982	78.7	0.44	0.6	0.20	314	0.86	7.7	2126	16381
1983	39.2	0.18	0.5	0.15	331	0.87	12.2	1058	12895
1984	29.1	0.08	0.4	0.07	342	0.88	13.2	786	10363
1985	61.6	0.10	0.6	0.09	371	0.89	9.0	1663	15015
1986	106.2	0.09	0.8	0.06	420	0.89	7.9	2867	22770
1987	113.9	0.11	0.9	0.09	416	0.88	7.6	3076	23363
1988	110.2	0.09	1.0	0.07	419	0.85	9.3	2975	27671
1989	131.0	0.12	1.0	0.07	498	0.87	7.3	3537	25993
1990	187.4	0.18	1.2	0.14	410	0.85	6.2	5060	31427
1991	153.9	0.36	0.9	0.16	422	0.90	5.9	4154	24677
1992	132.1	0.44	1.0	0.36	420	0.87	7.3	3566	26203
1993	96.6	0.11	0.5	0.08	396	0.90	5.2	2608	13553
1994	86.3	0.10	0.6	0.07	421	0.88	6.9	2330	15969
1995	138.7	0.12	0.9	0.09	420	0.91	6.6	3744	24733
1996	81.9	0.12	0.8	0.10	400	0.88	10.0	2212	22198
1997	59.3	0.10	0.8	0.11	431	0.89	14.0	1602	22345
1998	204.7	0.17	2.2	0.26	457	0.89	10.7	5527	58981
1999	201.7	0.16	2.1	0.12	424	0.93	10.5	5444	57265
2000	474.3	0.21	4.5	0.14	417	0.88	9.5	12805	121048
2001	336.4	0.09	4.5	0.10	452	0.92	13.3	9080	120786
2002	294.9	0.09	4.9	0.10	435	0.90	16.5	7961	131092
2003	455.5	0.12	5.8	0.08	426	0.91	12.7	12296	156244
2004	387.3	0.09	6.4	0.08	478	0.90	16.5	10456	172534
2005	300.4	0.07	6.0	0.07	468	0.93	20.0	8109	162195
2006	274.7	0.08	5.5	0.07	467	0.92	20.0	7415	148504
2007	250.1	0.06	4.9	0.06	472	0.93	19.8	6752	133446
2008	369.1	0.08	6.2	0.07	389	0.93	16.9	9964	168332
2009	392.0	0.11	6.5	0.10	375	0.93	16.5	10582	174873

Table B-5. SMAST large camera video survey mean densities for sea scallops 40+ mm SH.

Year	Stations	Area Surveyed km^2	Mean SH mm	SH Adj Area m^2	SH Adj Density sc/m^2	SE	95% CI
Georges Bank							
2003	929	**27906**	102.1	3.199	0.149	0.0117	0.0230
2004	935	**28430**	107.5	3.219	0.122	0.0145	0.0284
2005	902	**27844**	106.6	3.215	0.117	0.0127	0.0248
2006	939	**28276**	114.6	3.245	0.109	0.0116	0.0227
2007	912	**27813**	99.0	3.188	0.144	0.0160	0.0313
2008	910	**27227**	93.3	3.167	0.100	0.0087	0.0170
2009	899	**29079**	92.2	3.164	0.160	0.0175	0.0344
Mid Atlantic Bight							
Year	Stations	Area Surveyed km^2	Mean SH mm	SH Adj Area m^2	SH Adj Density sc/m^2	SE	95% CI
2003	804	**24664**	73.9	3.098	0.505	0.0835	0.1636
2004	840	**25591**	90.4	3.157	0.229	0.0224	0.0439
2005	864	**26547**	91.6	3.161	0.215	0.0254	0.0499
2006	897	**26918**	92.0	3.163	0.195	0.0193	0.0379
2007	941	**28739**	94.5	3.172	0.183	0.0163	0.0320
2008	931	**28184**	91.4	3.161	0.188	0.0187	0.0367
2009	928	**28647**	96.4	3.179	0.137	0.0085	0.0166

Table B-6. SMAST small camera video survey mean densities for sea scallops 40+ mm SH.

Year	Stations	Area Surveyed km^2	Mean SH mm	SH Adj Area m^2	SH Adj Density sc/m^2	Abundance	Biomass (mt)	SE	95% CI
					Georges Bank				
2003	904	27906	88.3	0.738	0.1698	4737032049	66669	0.0174	0.03
2004	921	28430	101.4	0.761	0.1256	3569624137	74432	0.0142	0.03
2005	902	27844	111.2	0.778	0.1001	2787348077	77929	0.0128	0.03
2006	916	28276	109.1	0.775	0.1412	3993072108	108805	0.0143	0.03
2007	901	27813	80.0	0.724	0.1974	5489504503	77729	0.022	0.04
2008	882	27227	99.4	0.758	0.1526	4153894290	102842	0.0189	0.04
2009	942	29079	96.1	0.752	0.1556	4525694473	94067	0.0186	0.04
					Mid-Atlantic Bight				
Year	Stations	Area Surveyed km^2	Mean SH mm	SH Adj Area m^2	SH Adj Density sc/m^2	Abundance	Biomass (mt)	SE	95% CI
2003	799	24664	58.6	0.688	0.7063	17419973913	81353	0.1427	0.28
2004	829	25591	84.7	0.732	0.2319	5935561328	69252	0.0263	0.05
2005	860	26547	87.2	0.737	0.2181	5790580803	81756	0.0267	0.05
2006	872	26918	93.4	0.747	0.2049	5516773301	88323	0.0216	0.04
2007	931	28739	90.4	0.742	0.2204	6333997245	88941	0.0213	0.04
2008	913	28184	90.7	0.743	0.2160	6086579306	103164	0.0207	0.04
2009	928	28647	98.1	0.755	0.1260	3608213579	71936	0.0091	0.02

Table B-7. Comparison of various estimates of New Bedford scallop dredge efficiencies.

Source	Area	Gear	Method	Estimates	cv
NEFSC(2010)	MA + GB sand	Lined survey	Paired camera/dredge comarisons	0.44	0.01
NEFSC(2010)	GB gravel/cobble	Lined survey	Paired camera/dredge comarisons	0.38	0.03
NEFSC(2007)	ALL	Lined survey	Compairson of video,dredge surveys	0.38	0.10
NEFSC(2007)	Georges Bank	Lined survey	Compairson of video,dredge surveys	0.37	0.18
NEFSC(2007)	Mid-Atlantic	Lined survey	Compairson of video,dredge surveys	0.4	0.07
NEFSC(2004)	Georges Bank	Lined survey	Compairson of video,dredge surveys	0.33	
NEFSC(2004)	Mid-Atlantic	Lined survey	Compairson of video,dredge surveys	0.46	
NEFSC(2001)	Georges Bank	Commercial	Depletion	0.38-0.81	0.19-0.98
NEFSC(2001)	Mid-Atlantic	Commercial	Depletion	0.59-0.75	0.1-0.72
Gedamke et al. 2005	Georges Bank CAII	Commercial	Index removal	0.41-0.54	
Gedamke et al. 2004	Georges Bank CAII	Commercial	Depletion	0.35-0.525	
Caddy 1971	Canada	Commercial	Dredge mounted camera	0.17	

Table B-8. CASA model estimates and standard errors for fully recruited sea scallop fishing mortality, July 1 abundance 40+mm SH, and July 1 biomass 40+ mm SH.

		Georges Bank							MidAtlantic						Total	
Year	Full_F	SE	Abundance (millions)	SE	Biomass (mt meats)	SE	Full_F	SE	Abundance (millions)	SE	Biomass (mt meats)	SE	Full_F	SE	Abundance (millions)	SE
1975	0.11	0.02	1148	56	20780	1038	0.59	0.09	591	34	6503	386	0.21	0.09	1739	66
1976	0.20	0.04	1419	60	24705	1112	1.00	0.16	787	33	7931	491	0.38	0.16	2205	69
1977	0.33	0.05	1115	52	24522	1056	0.53	0.07	772	30	9933	487	0.39	0.23	1886	60
1978	0.39	0.06	1260	51	21973	920	1.05	0.15	567	21	9690	443	0.57	0.3	1827	55
1979	0.53	0.08	878	40	17822	762	1.07	0.20	364	15	7678	364	0.63	0.44	1242	43
1980	0.47	0.08	1060	43	14970	628	0.35	0.05	343	16	6365	347	0.44	0.34	1403	45
1981	0.62	0.09	747	34	12579	533	0.13	0.03	403	18	6754	364	0.48	0.44	1151	38
1982	0.83	0.13	808	35	9505	425	0.25	0.04	442	21	7401	386	0.58	0.46	1250	41
1983	0.71	0.11	573	30	7680	393	0.53	0.07	497	25	6987	417	0.64	0.41	1070	39
1984	0.42	0.08	565	34	7364	442	0.80	0.12	536	31	6062	459	0.58	0.25	1101	46
1985	0.51	0.10	610	42	7840	528	0.75	0.13	744	40	6346	506	0.61	0.3	1354	58
1986	0.88	0.21	984	60	8481	542	0.57	0.09	977	47	8704	556	0.72	0.41	1962	76
1987	0.76	0.16	1096	66	9988	596	1.20	0.17	1171	49	9340	585	0.96	0.43	2267	82
1988	0.83	0.18	1251	77	11321	686	0.90	0.12	`	49	10365	558	0.86	0.44	2399	91
1989	0.64	0.13	1415	81	13453	736	1.14	0.15	1147	42	9852	534	0.85	0.39	2562	91
1990	1.11	0.21	1369	74	12791	678	0.96	0.11	1018	36	9747	418	1.05	0.63	2387	82
1991	1.53	0.28	1486	68	10725	475	1.07	0.10	705	26	8026	327	1.32	0.8	2191	73
1992	1.72	0.25	783	36	7056	303	1.10	0.12	468	24	5426	298	1.47	1.01	1251	43
1993	1.19	0.21	553	32	4868	279	0.86	0.14	894	38	5634	319	1.05	0.66	1448	49
1994	0.31	0.07	531	36	5719	394	1.37	0.18	1137	40	8027	360	0.87	0.18	1668	53
1995	0.16	0.03	1003	48	9878	553	1.08	0.11	965	34	8785	361	0.62	0.1	1968	59
1996	0.33	0.07	1201	53	15406	727	0.74	0.08	647	31	8167	411	0.53	0.18	1849	62
1997	0.28	0.07	1305	62	20141	885	0.47	0.06	690	44	7850	528	0.35	0.18	1995	76
1998	0.22	0.06	1924	82	27276	1022	0.53	0.10	1695	82	11858	716	0.31	0.16	3619	116
1999	0.54	0.13	2008	87	33163	1211	0.49	0.09	2872	106	23689	1043	0.51	0.23	4881	137
2000	0.48	0.12	3129	99	41066	1410	0.48	0.08	3523	112	37324	1326	0.48	0.14	6652	149
2001	0.26	0.05	3294	95	53064	1704	0.54	0.07	3766	107	45795	1433	0.43	0.11	7061	143
2002	0.23	0.05	2819	88	62370	1994	0.61	0.08	3427	100	48798	1449	0.41	0.12	6246	133
2003	0.17	0.04	2945	96	69416	2294	0.68	0.08	4174	115	48756	1397	0.42	0.1	7119	150
2004	0.10	0.02	2708	96	74629	2603	0.87	0.09	3703	112	50029	1468	0.38	0.07	6411	147
2005	0.18	0.03	2571	103	73828	2862	0.84	0.14	3609	131	49027	1728	0.37	0.13	6180	167
2006	0.38	0.06	2128	108	62768	3090	0.35	0.06	3805	166	56405	2377	0.37	0.23	5933	198
2007	0.25	0.05	2364	151	53650	3472	0.55	0.09	3853	209	61784	3260	0.40	0.14	6217	258
2008	0.19	0.04	2769	204	55508	4234	0.54	0.10	4509	313	63983	4518	0.37	0.11	7278	374
2009	0.18	0.05	3453	294	62470	5341	0.60	0.13	3993	352	67233	6460	0.38	0.11	7446	458

Table B-9. Biological reference points from the previous and current sea scallop assessments.

Reference point	SARC-45, whole Stock	Updated		
		GBK	MAB	Whole stock
F_{MSY}	--	0.21	0.47	0.38
B_{MSY} (July 1, 40+ mm SH)	--	41,468	86,330	125,358
$B_{Threshold}$=1/2 B_{MSY}	--	20,734	43,165	62,679
MSY	--	6,410	19,040	24,975
F_{MAX} (SYM)		0.295	0.835	0.48
F_{MAX} (CASA)	0.29	0.23	0.375	0.30
B_{MSY} proxy (CASA) (Jan. 1, 40+ mm)	108,628	--	--	127,000
$B_{Threshold}$=1/2 B_{MSY} proxy	54,314	--	--	63,500

Figures

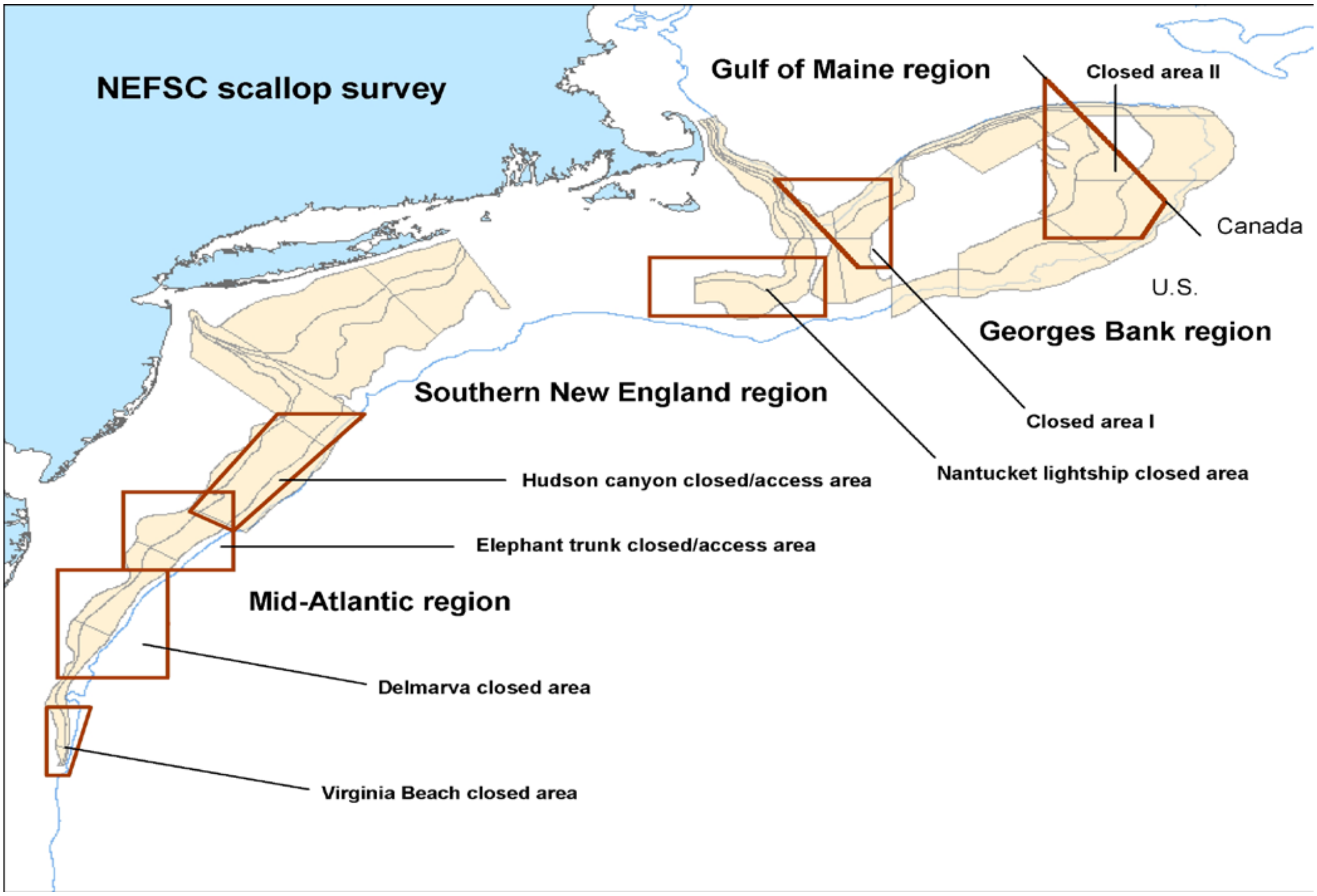


Figure B-1. Map of NEFSC sea scallop survey areas (yellow, with stratum boundaries shown) and the closed or rotational access areas (bounded by dark red lines).

(a) (b)

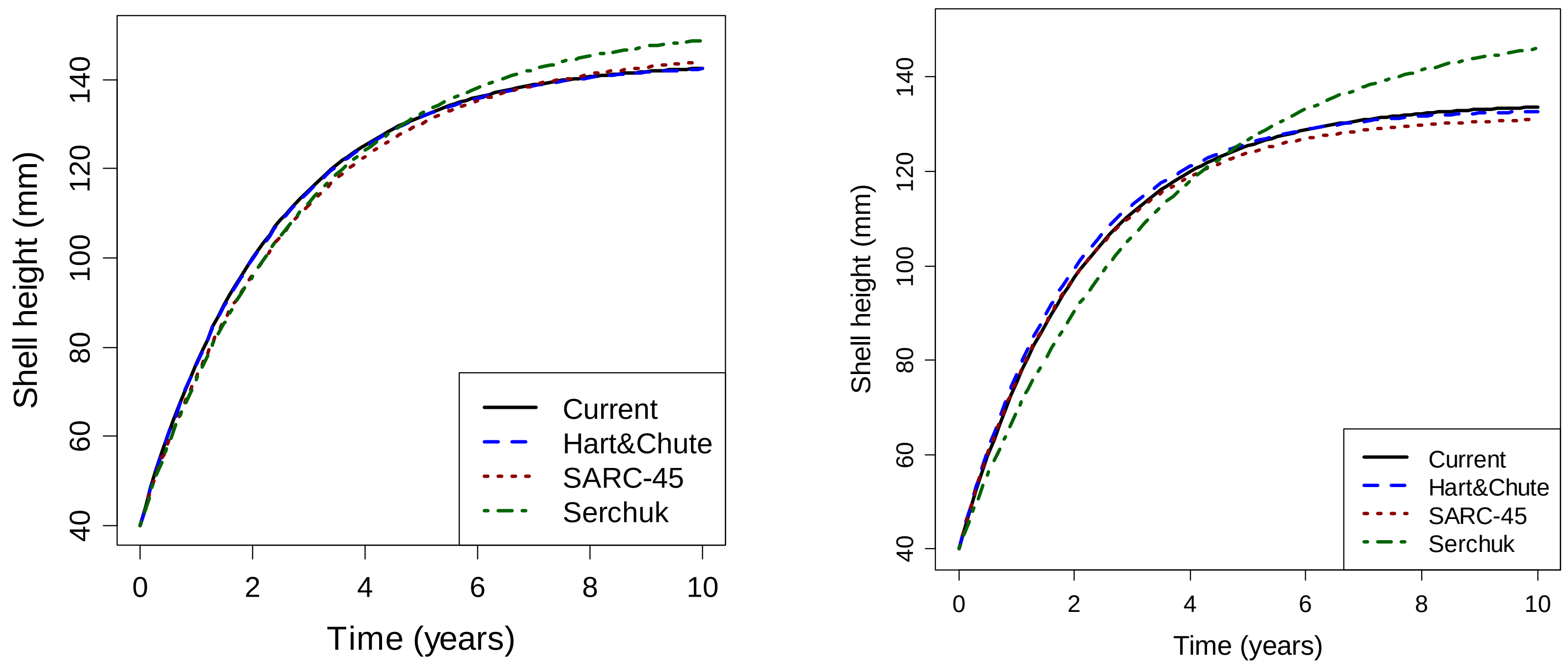
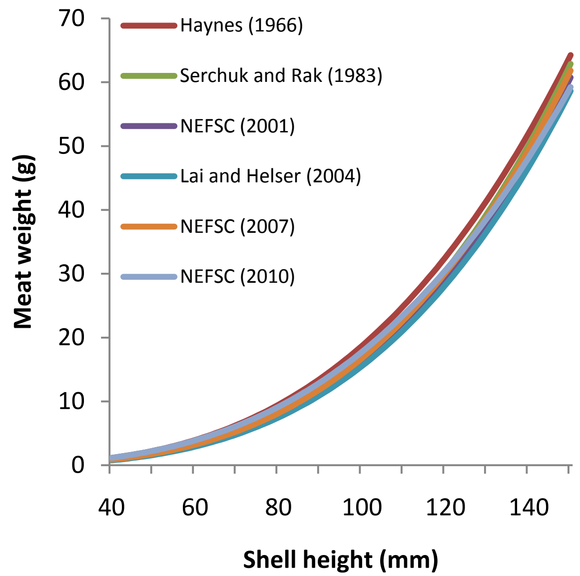


Figure B-2. Comparison of growth curves for a scallop with starting shell height of 40 mm in (a) Georges Bank and (b) Mid-Atlantic.

(a)



(b)

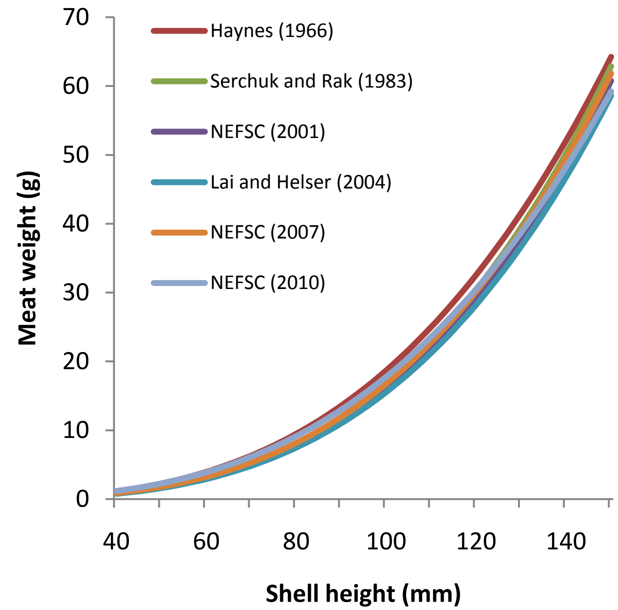


Figure B-3. Comparison of new shell height/meat weight relationships (calculated ignoring depth effects) for (a) Georges Bank and (b) Mid-Atlantic with other shell height/meat weight curves.

(a) (b)

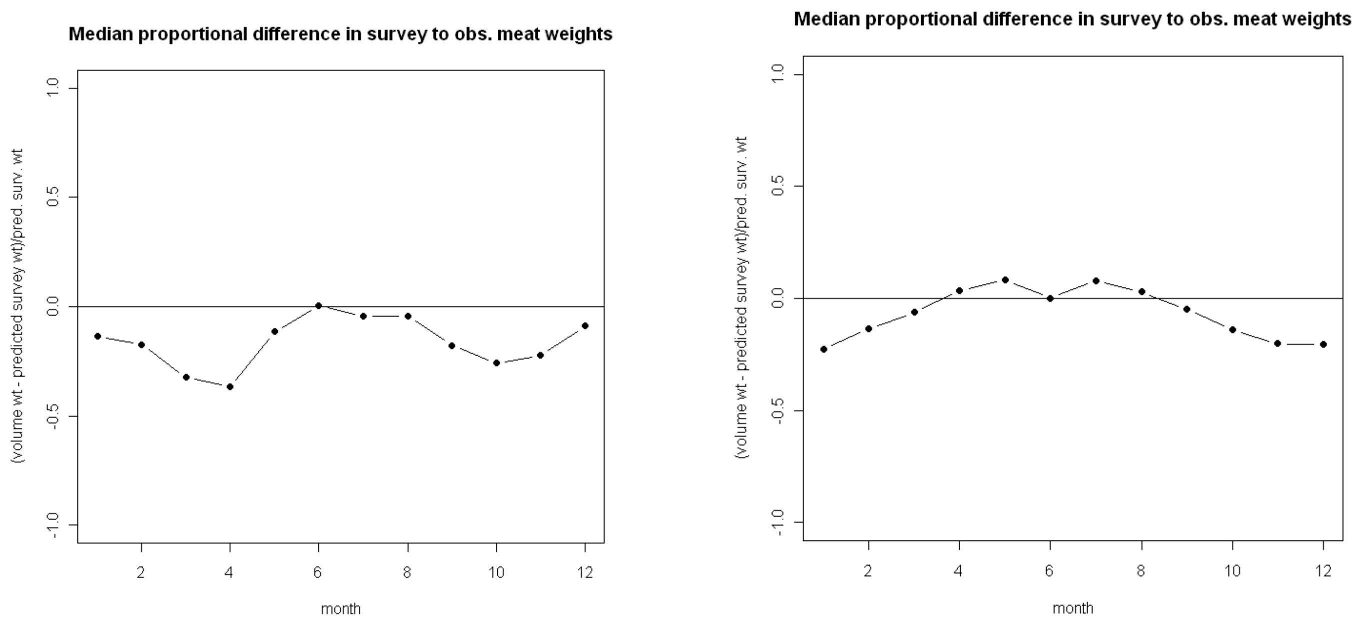


Figure B-4. Seasonal anomalies in shell height/meat weight relationships relative to that estimated from R/V data for (a) Georges Bank and (b) Mid-Atlantic Bight.

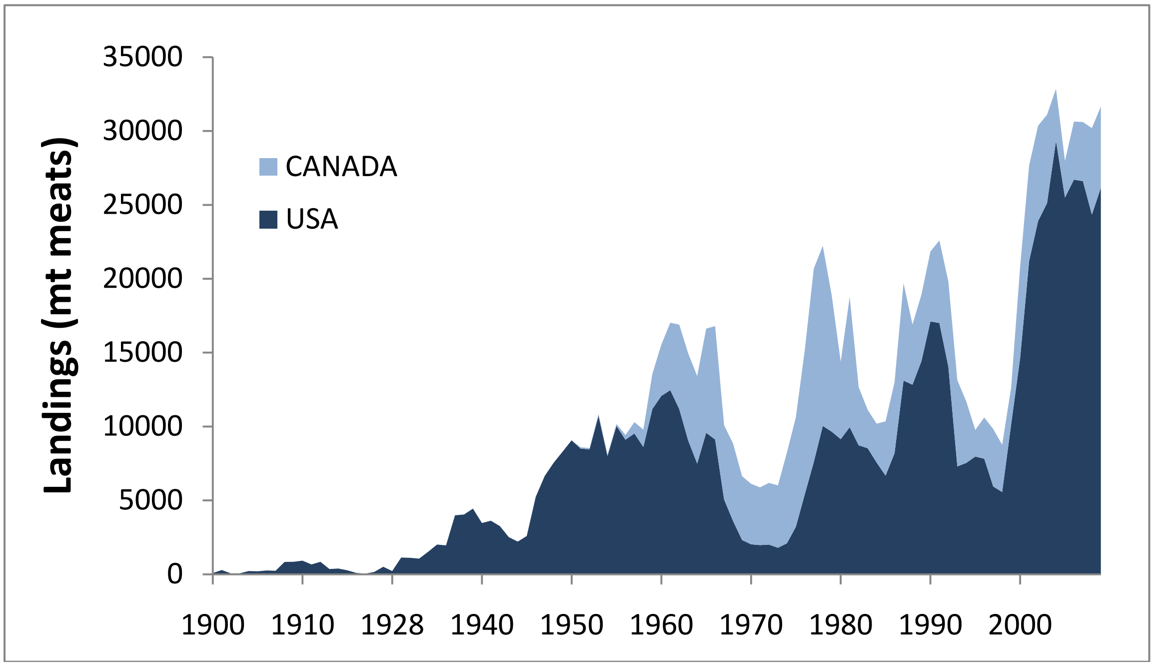


Figure B-5. Long-term sea scallop landings in NAFO areas 5 and 6 (U.S. and Canadian Georges Bank).

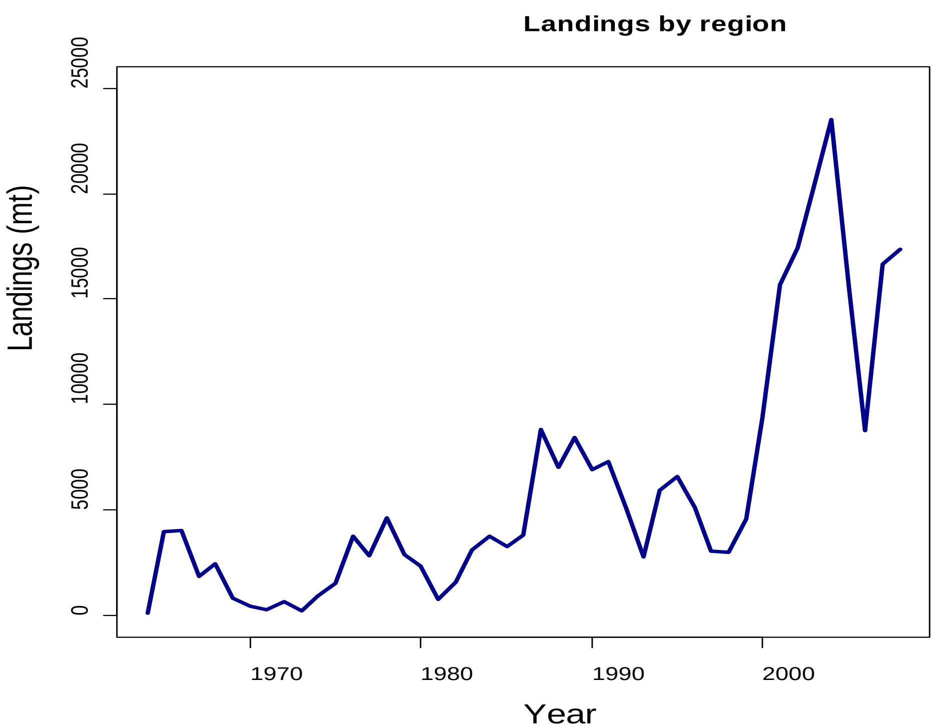
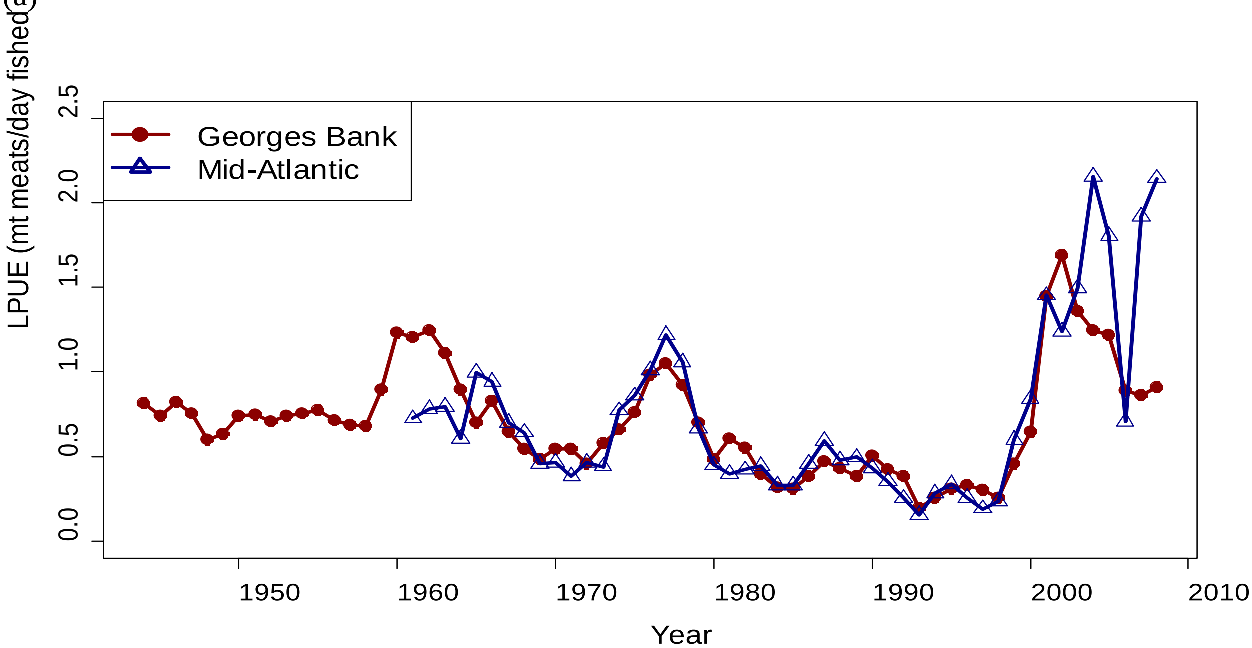


Figure B-6. U.S. sea scallop landings (mt meats) by region.

(a)



(b)

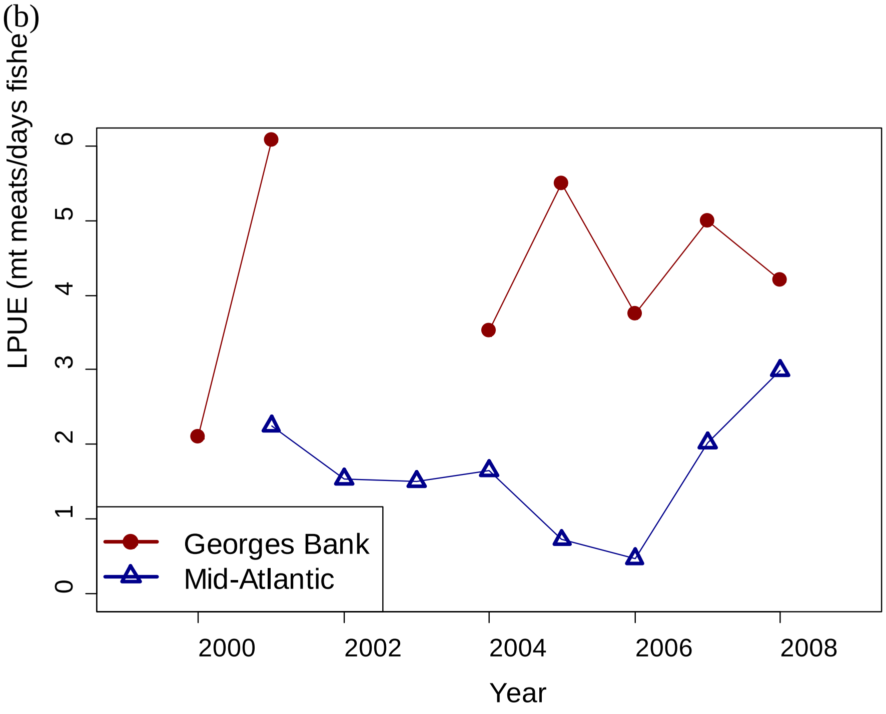
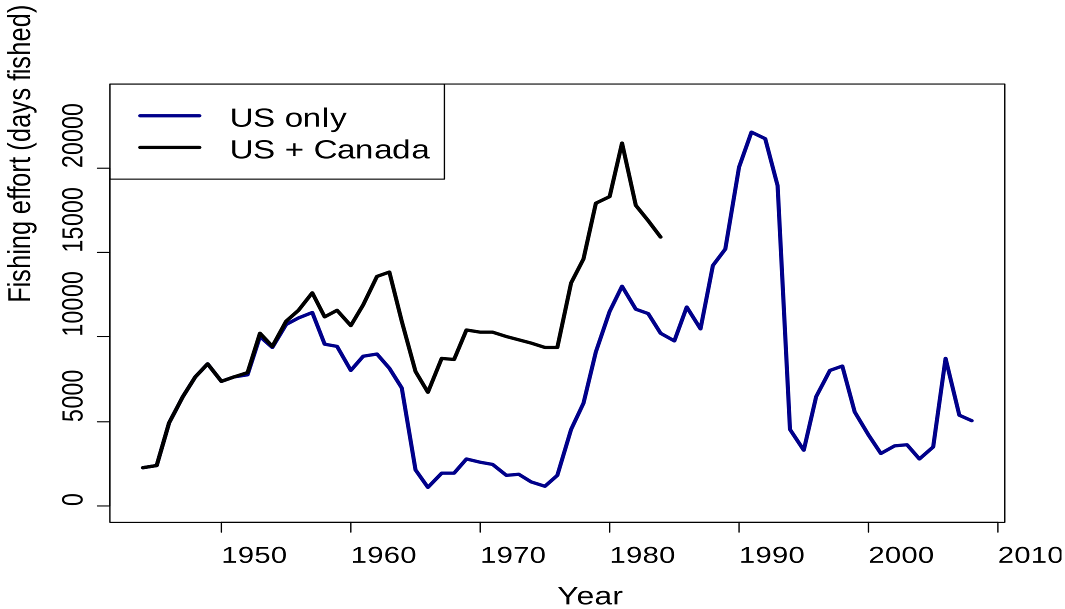


Figure B-7. Landings per day fished in (a) “open” areas, and (b) special access areas.

(a)



(b)

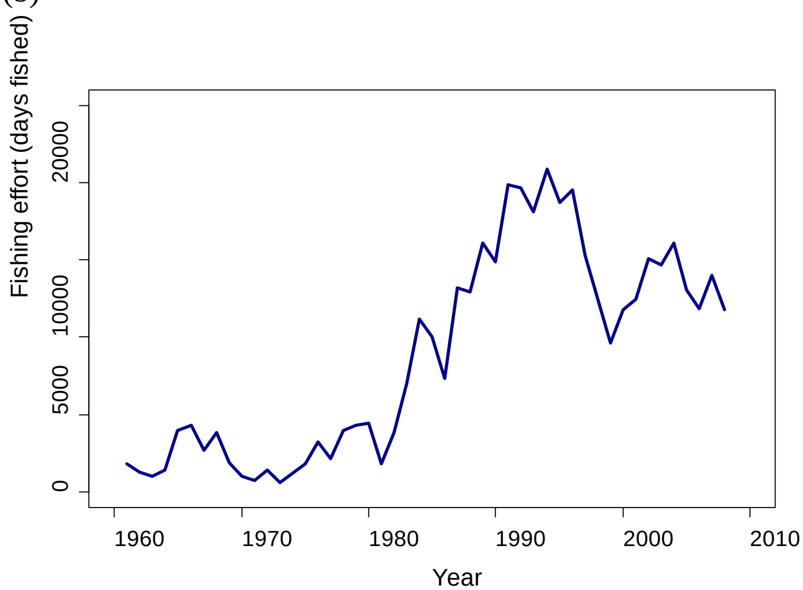


Figure B-8. Days fished in the sea scallop fishery in (a) Georges Bank and (b) Mid-Atlantic

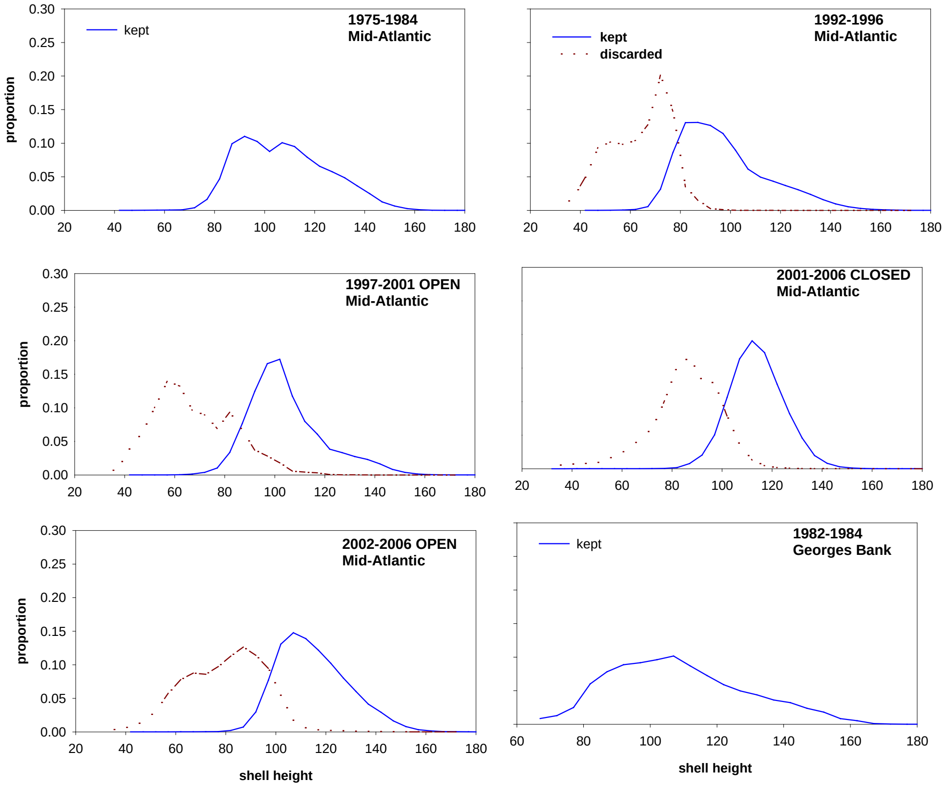


Figure B-9. Shell heights of commercial kept (solid line) and discarded (dashed line) sea scallops, from port sampling (1975-1984) and sea sampling (1992-2009).

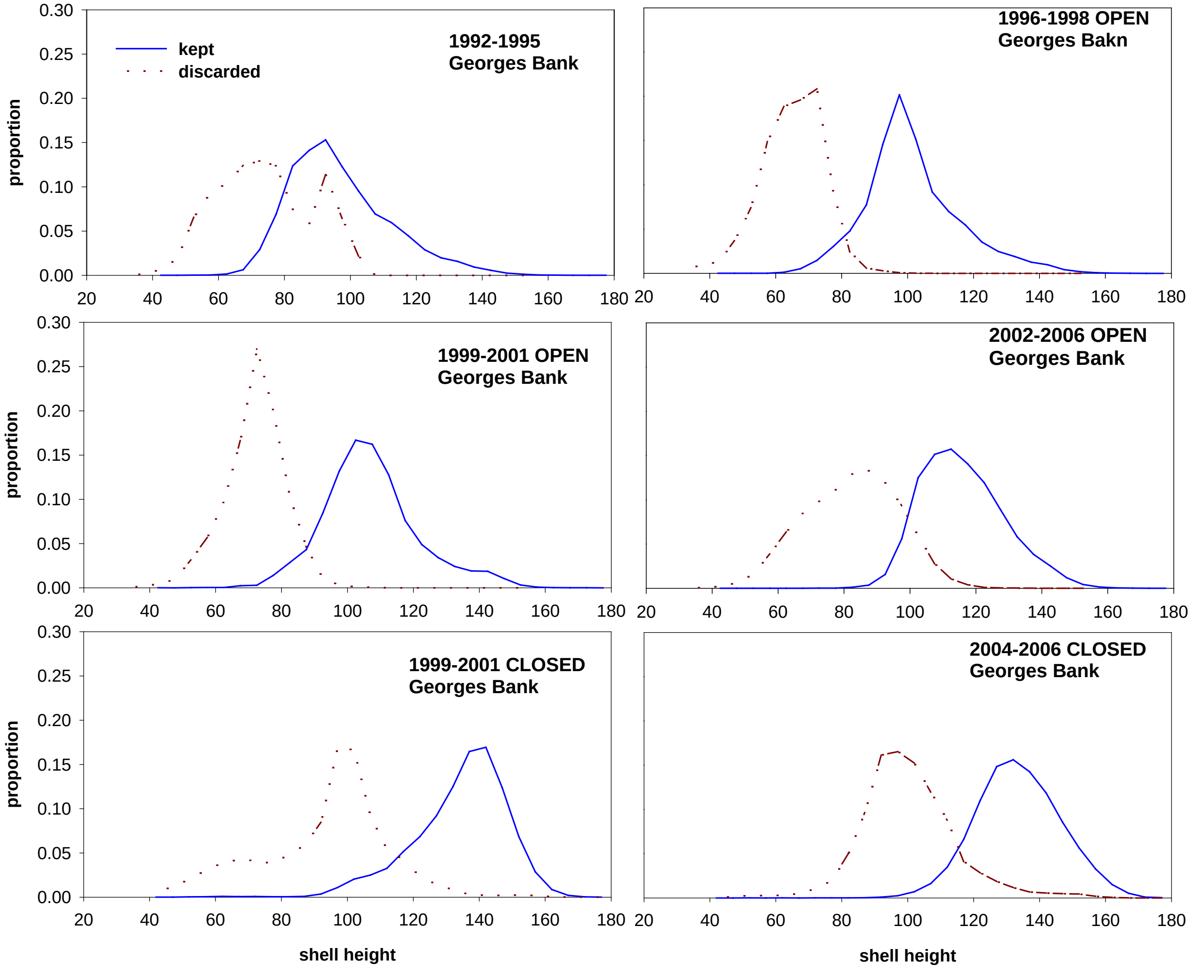


Figure B-9 continued

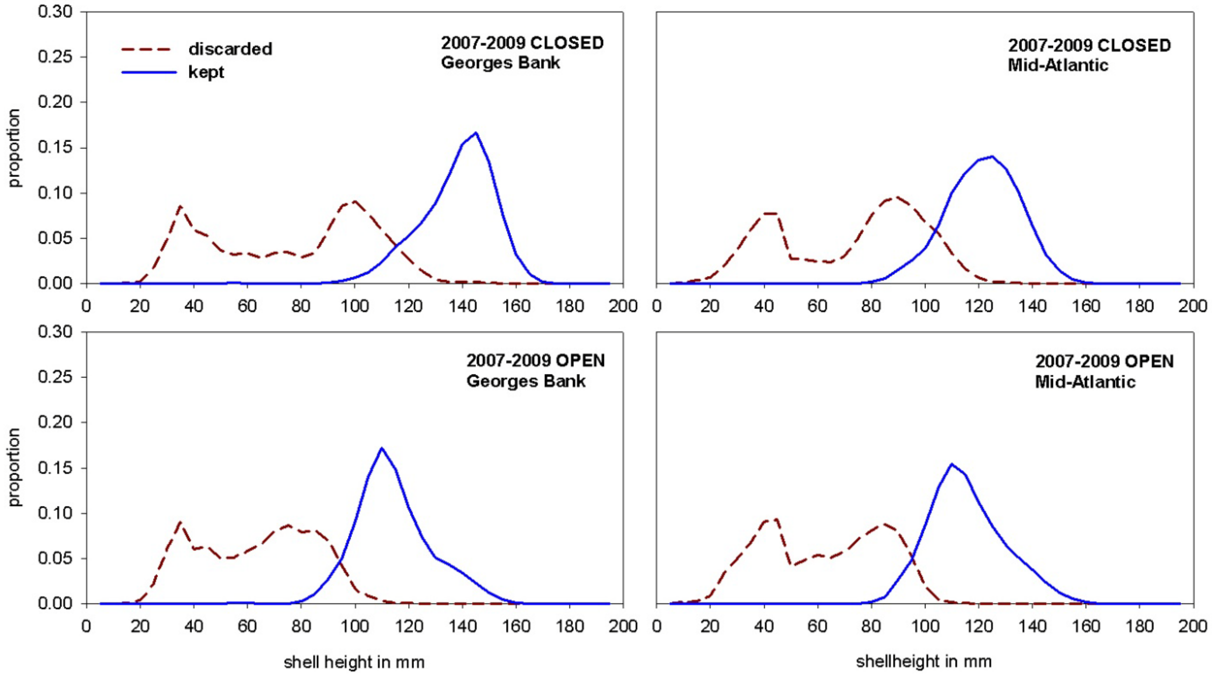


Figure B-9 continued

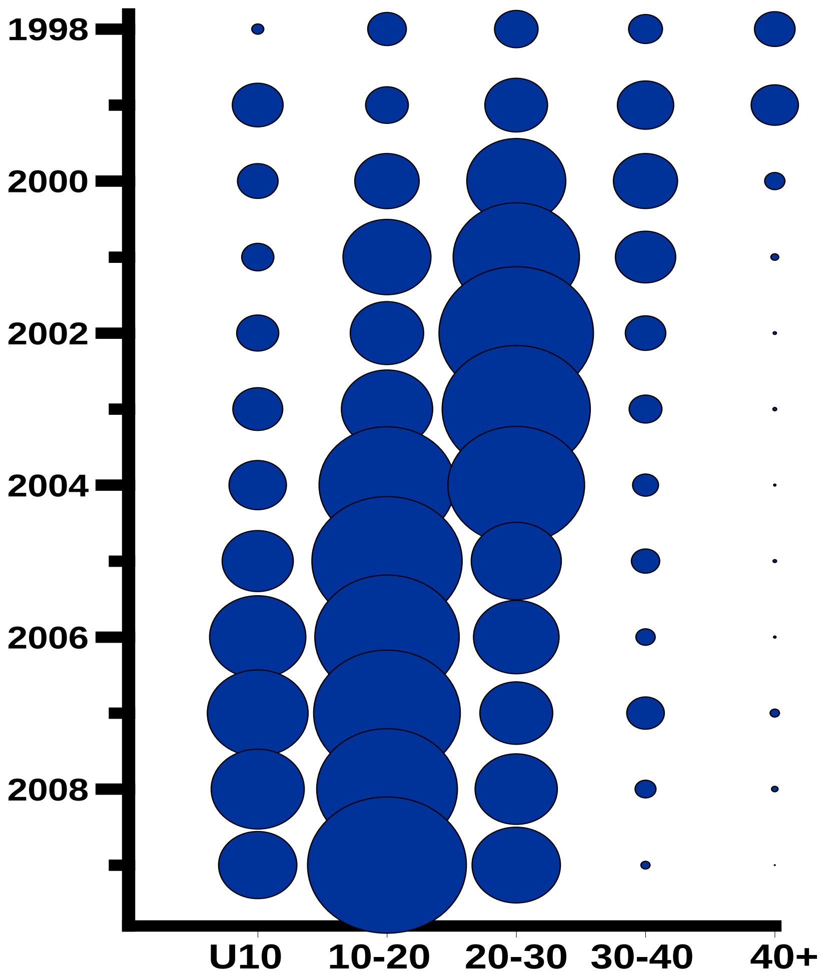


Figure B-10. Commercial landings by meat count category (number of meats per pound, U10 = less than 10 meats per pound).

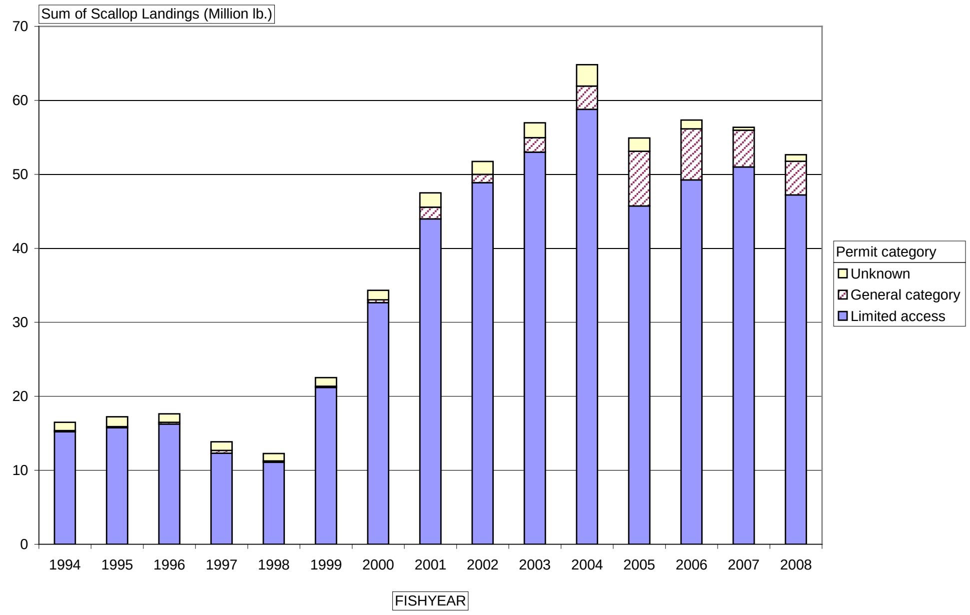


Figure B-11. Landings by permit category and fishing year (fishing year starts March 1).

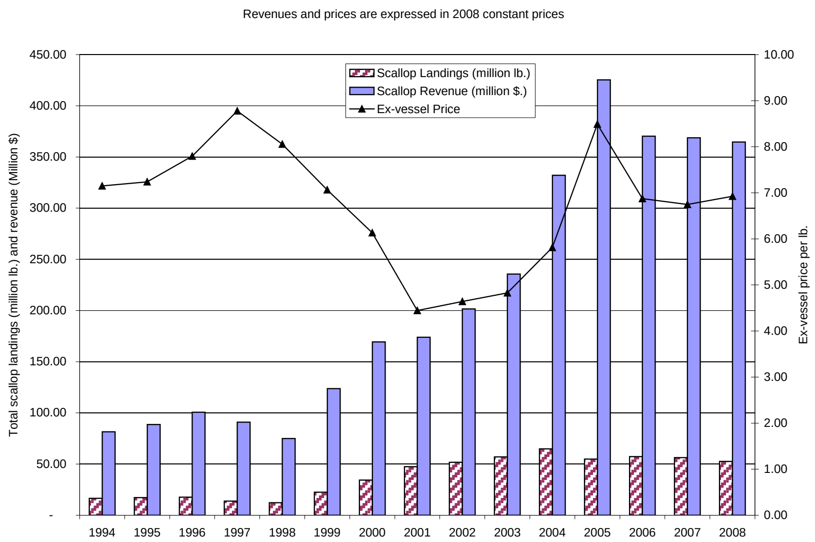


Figure B-12. Trends in scallop landings, revenue and ex-vessel prices by fishing year.

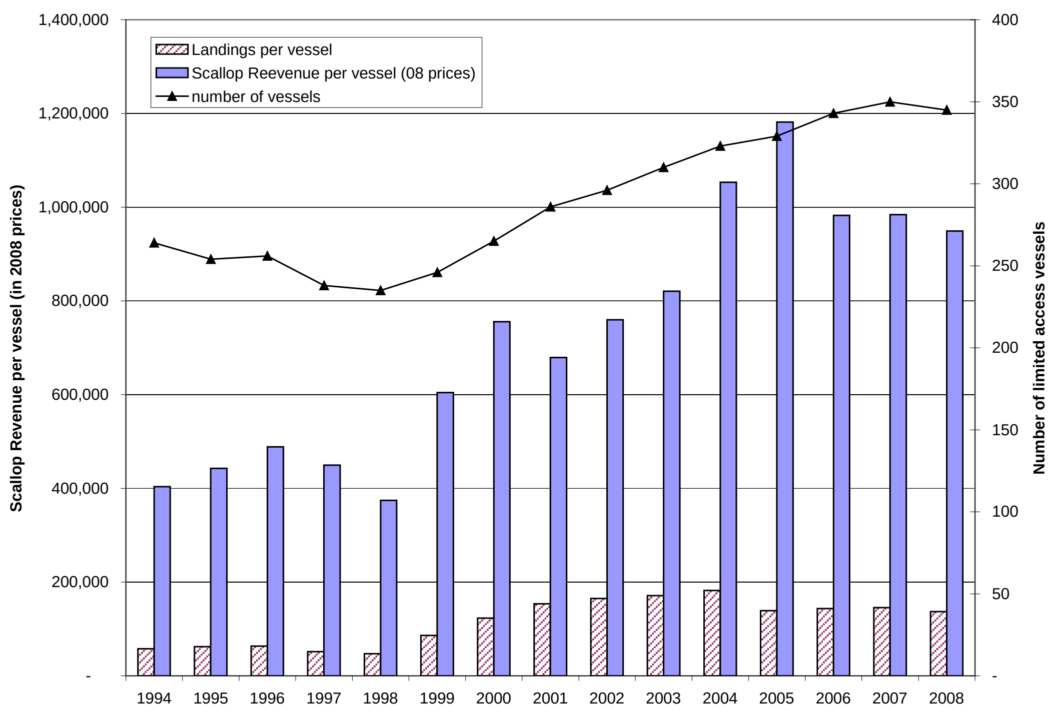


Figure B-13. Trends in average scallop landings and revenue per full time vessel and number of active limited access vessels.

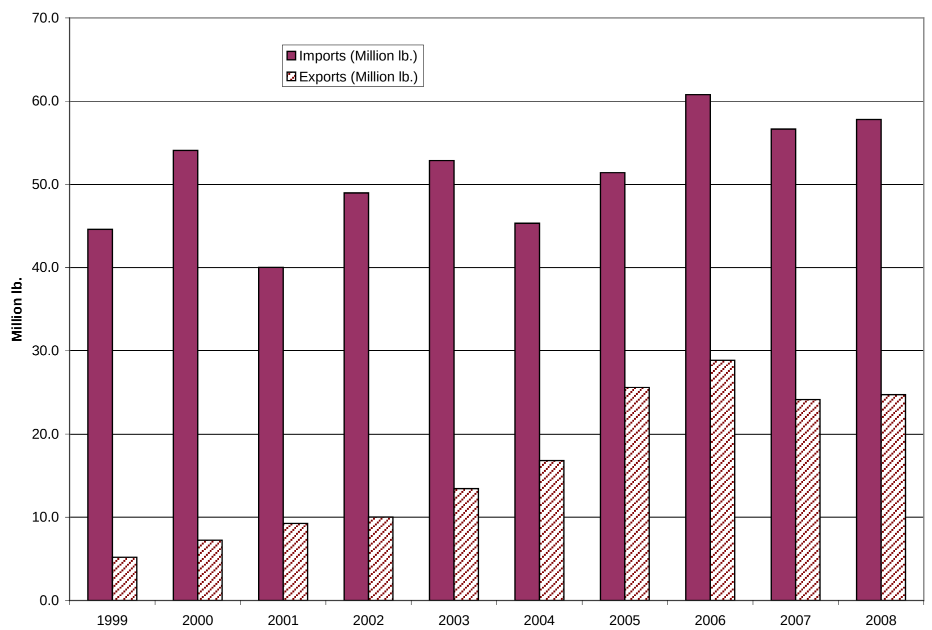
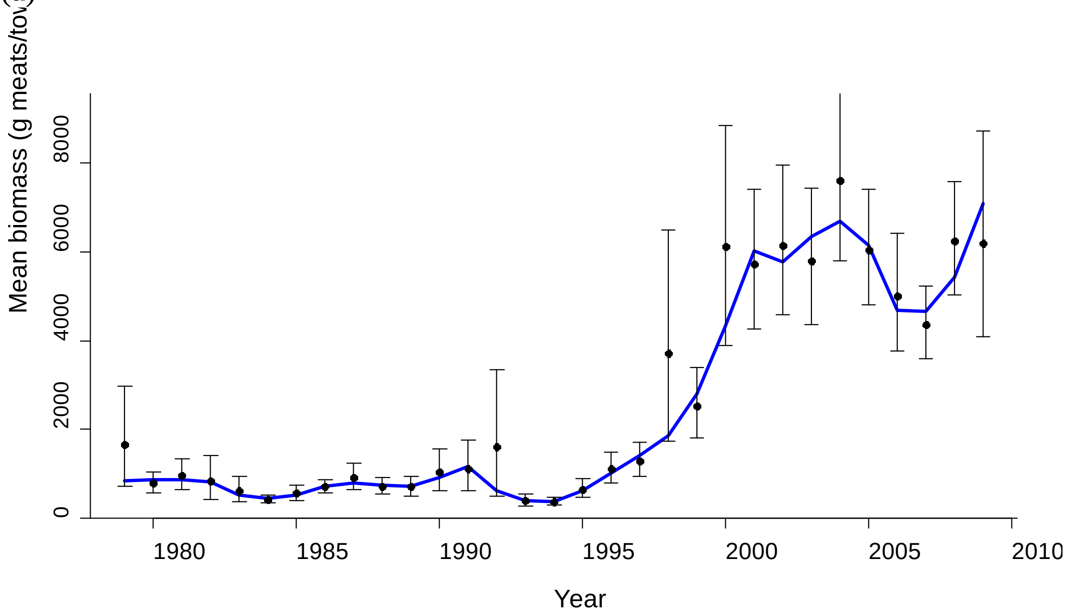


Figure B-14. Scallop exports and imports (includes other scallop species).

(a)



(b)

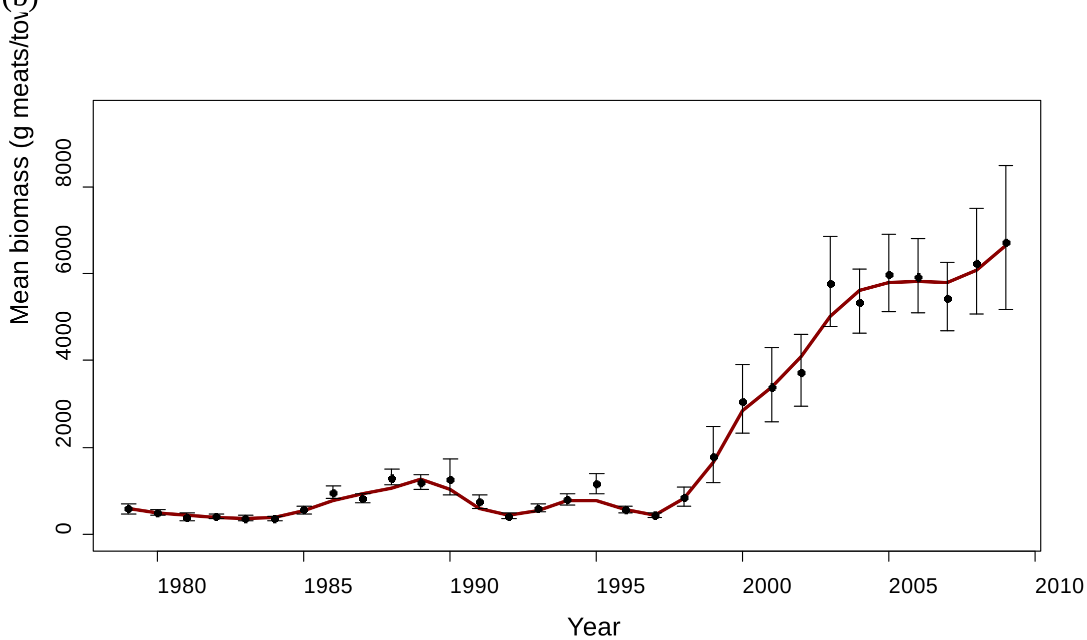
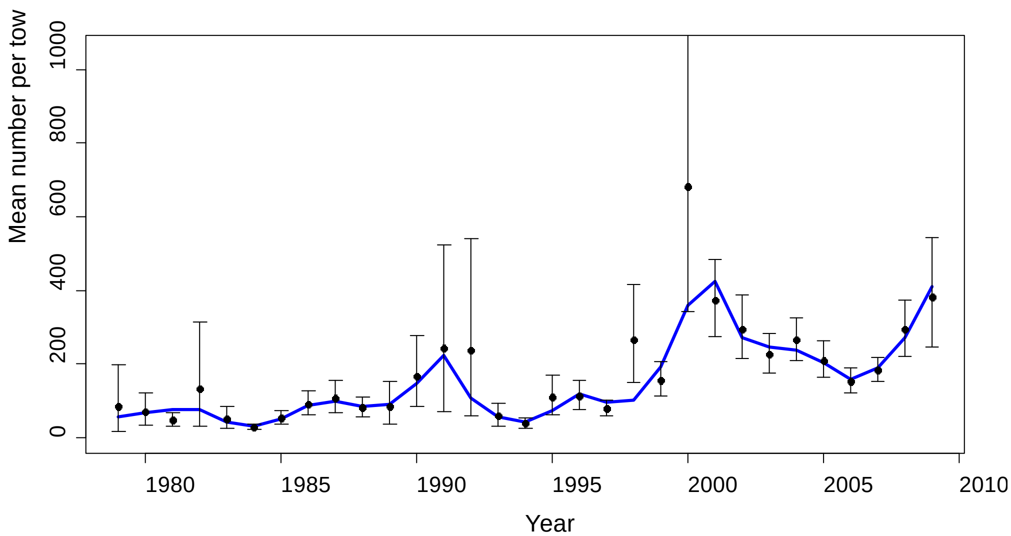


Figure B-15. NEFSC lined dredge sea scallop survey biomass indices in (a) Georges Bank and (b) Mid-Atlantic. 95% confidence intervals and inverse variance weighted lowess smoothers (lines, span = 0.25) are also shown.

(a)



(b)

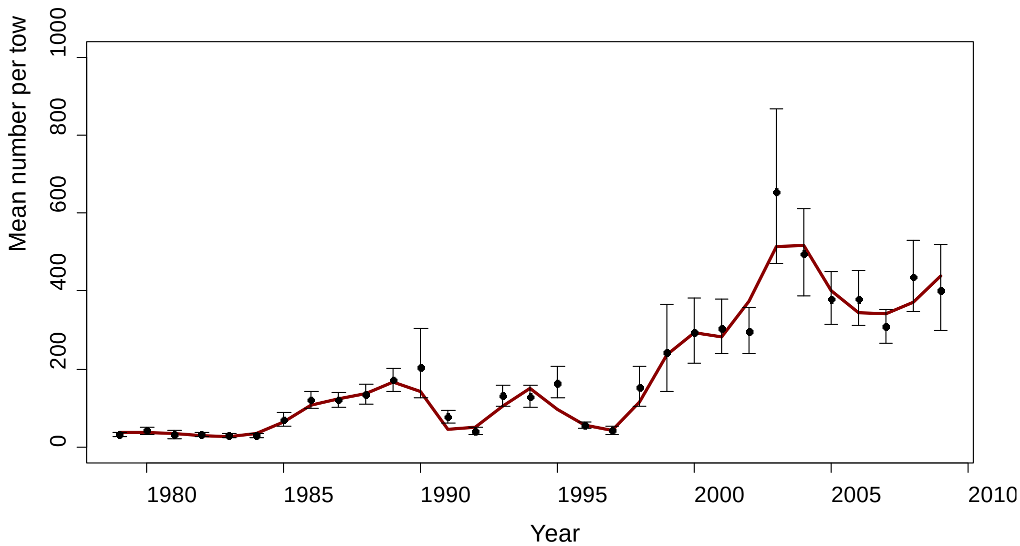
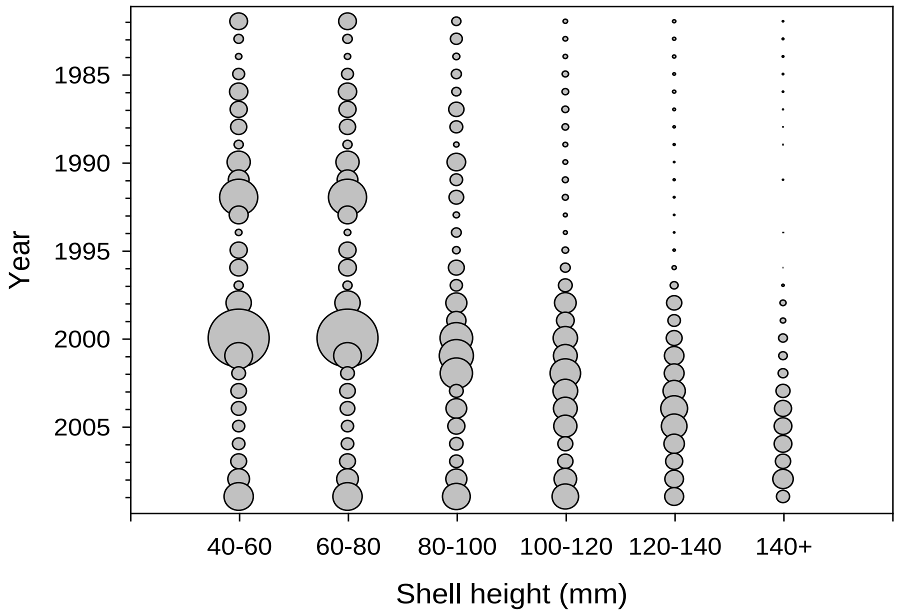


Figure B-16. NEFSC lined dredge sea scallop survey abundance indices in (a) Georges Bank and (b) Mid-Atlantic. 95% confidence intervals and inverse variance weighted lowess smoothers (lines, span = 0.25) are also shown.

(a)



(b)

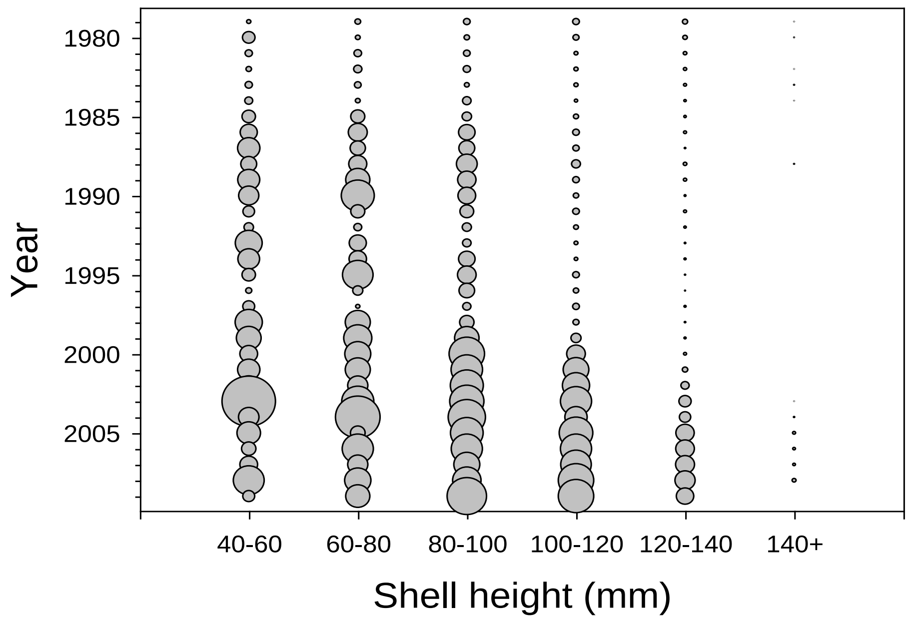


Figure B-17. Numbers of scallops by shell height group for (a) Georges Bank and (b) Mid-Atlantic, based on the NEFSC lined dredge survey.

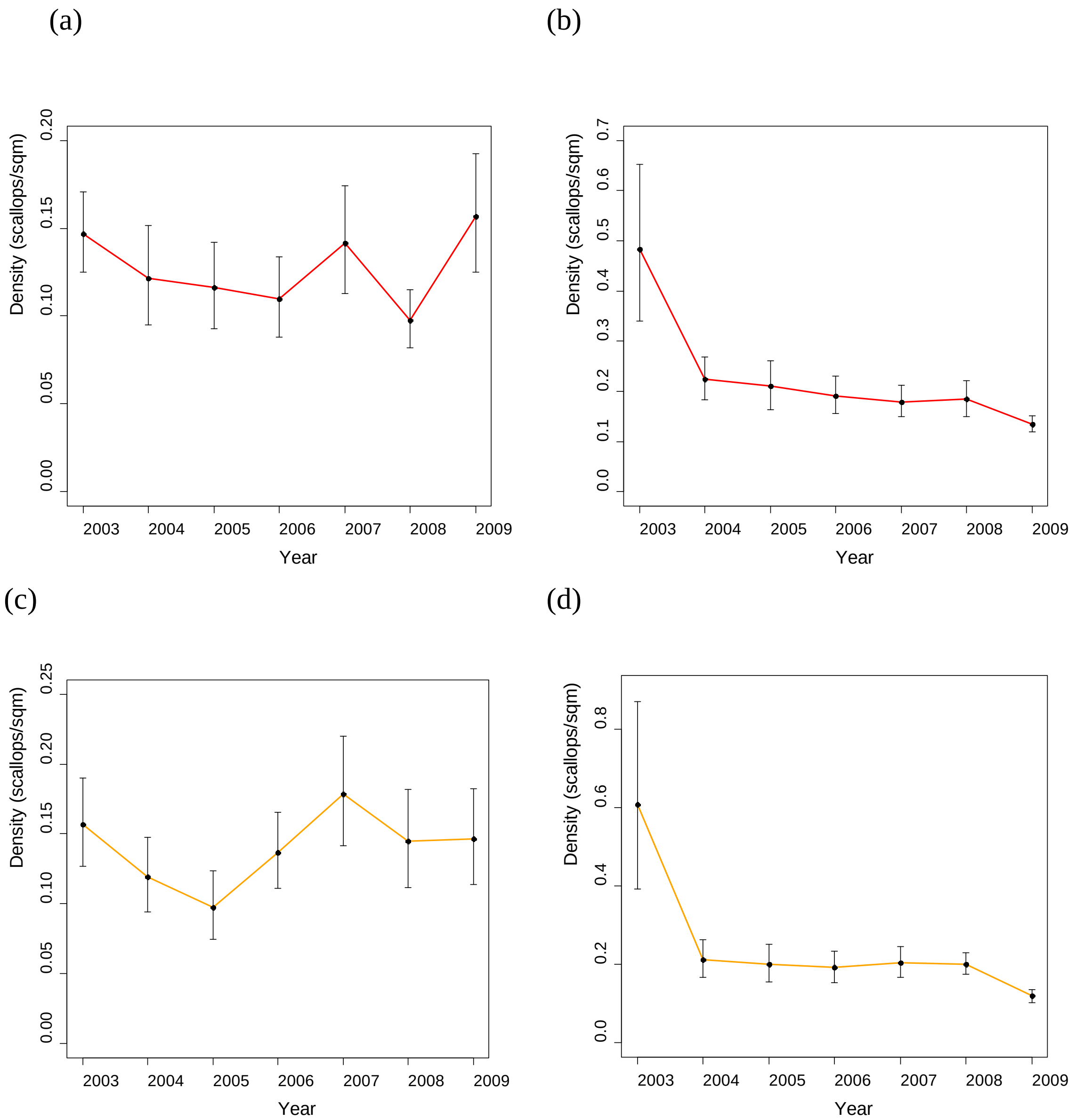
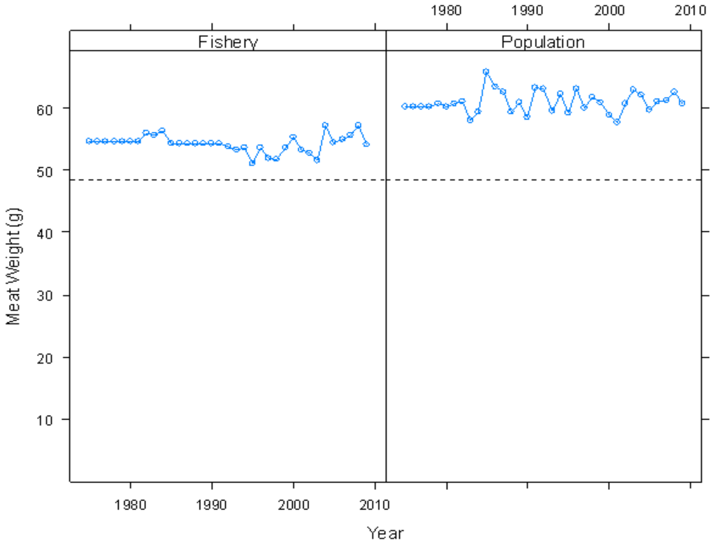


Figure B-18. Sea scallop density estimates from (a-b) the large video camera, and (c-d) the small video camera, in (a) and (c) Georges Bank and (b) and (d) the Mid-Atlantic. 95% confidence intervals are also shown.

(a)



(b)

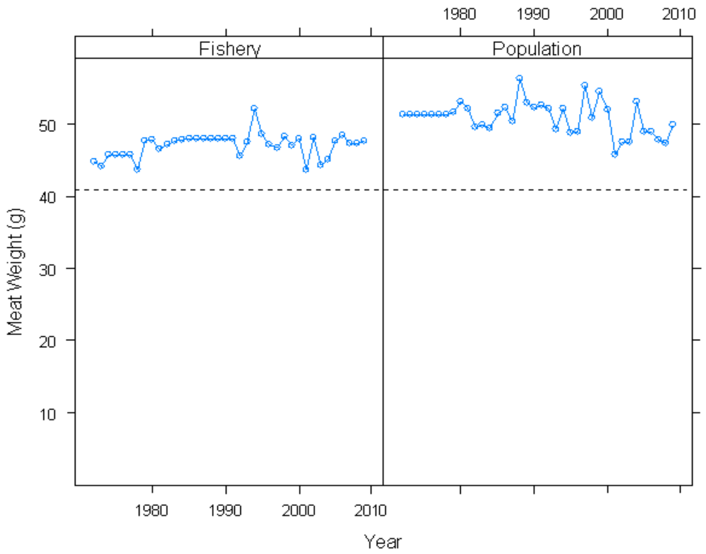


Figure B-19. Plus group meat weights for the population and the fishery in (a) Georges Bank and (b) Mid-Atlantic. Plus groups represent >140 mm SH in Geroges Bank and >130mm SH in the Mid-Atlantic.

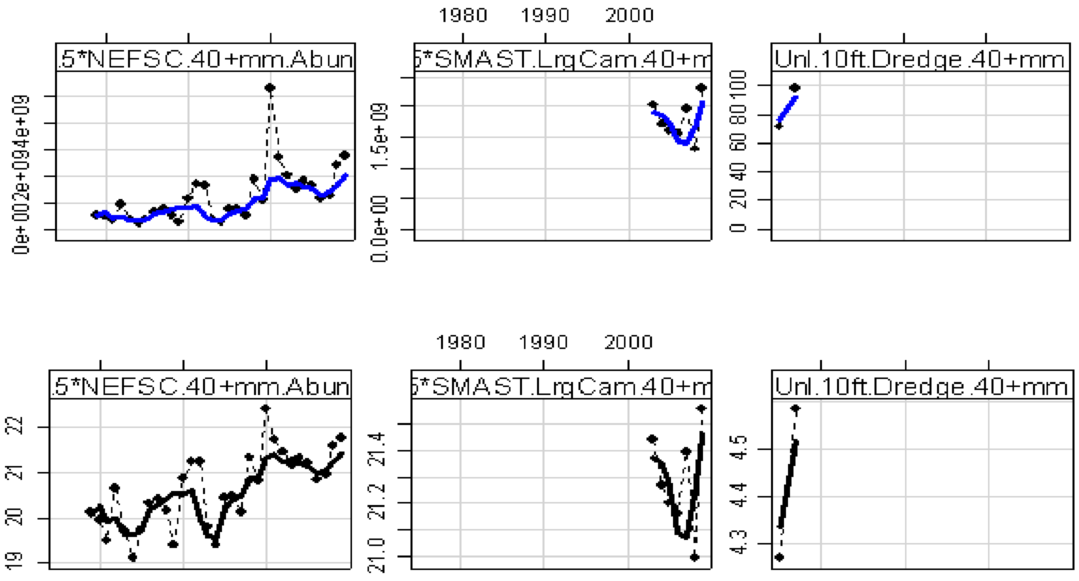


Figure B-20. Comparison between survey trend data (solid circles) and corresponding model estimates (lines) for the NEFSC lined dredge survey, the SMAST large camera survey and the NEFSC unlined dredge survey. Results are shown on a linear scale (top) and a log scale (bottom).

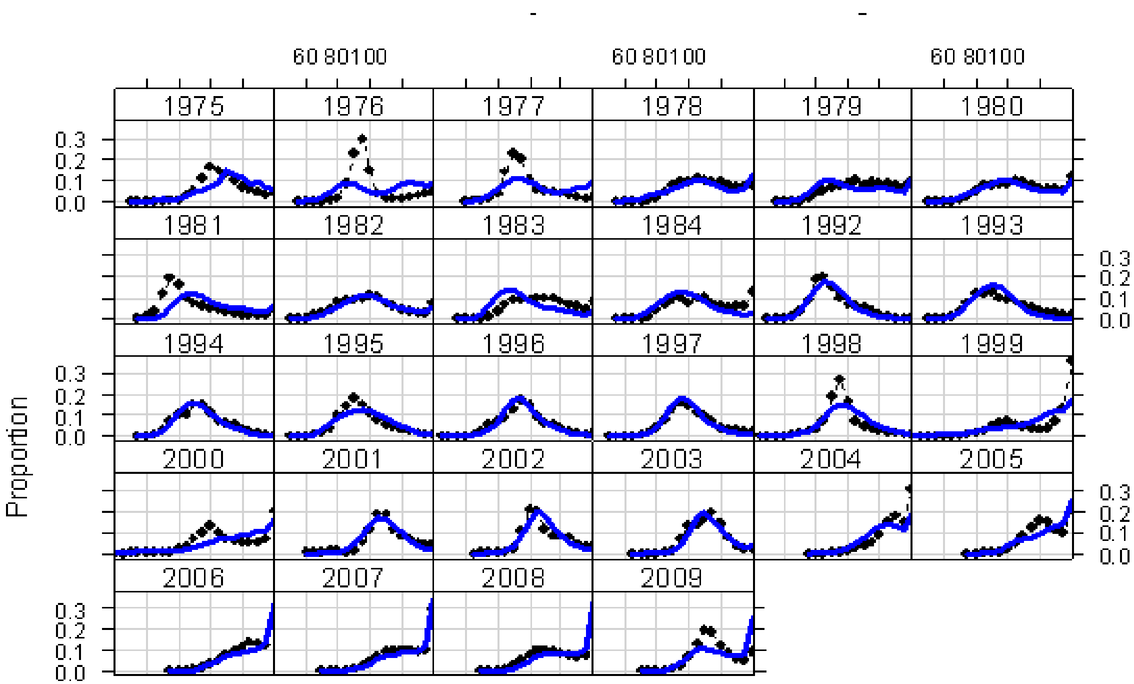


Figure B-21. Comparison of fishery shell height proportions (solid circles) and model estimated fishery shell height proportions (lines) for Georges Bank.

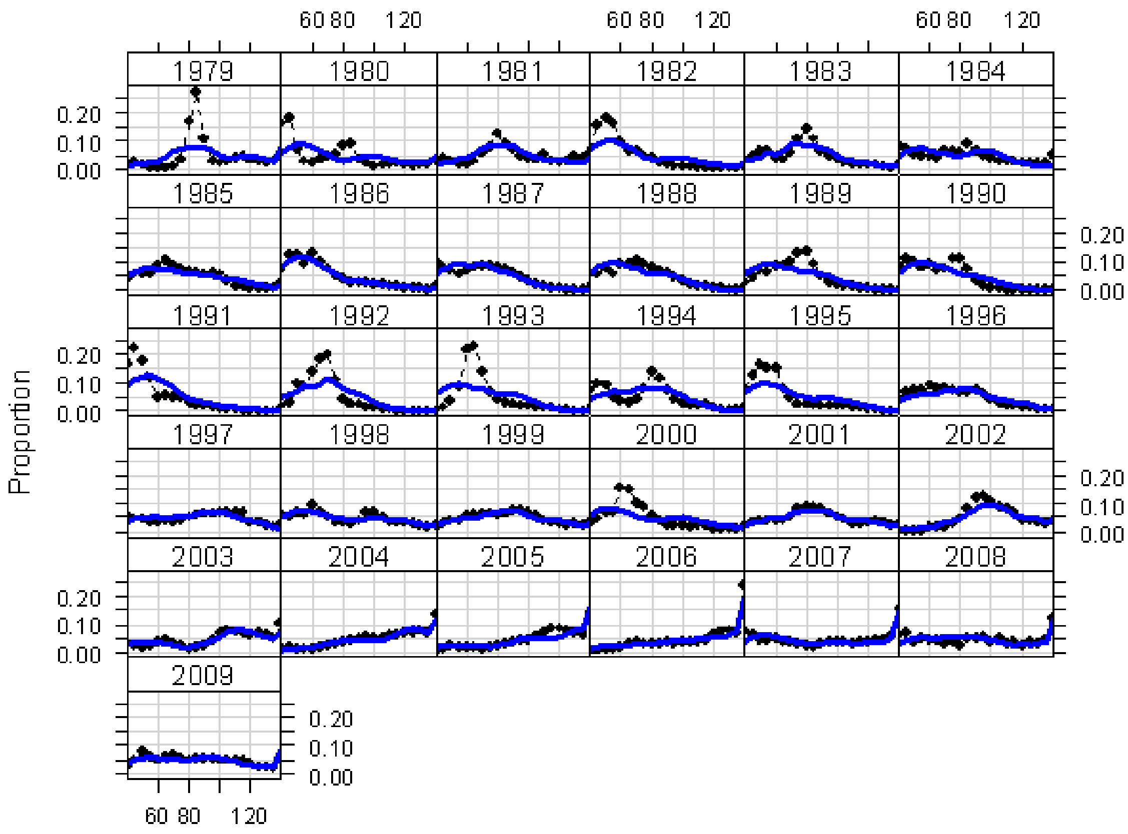
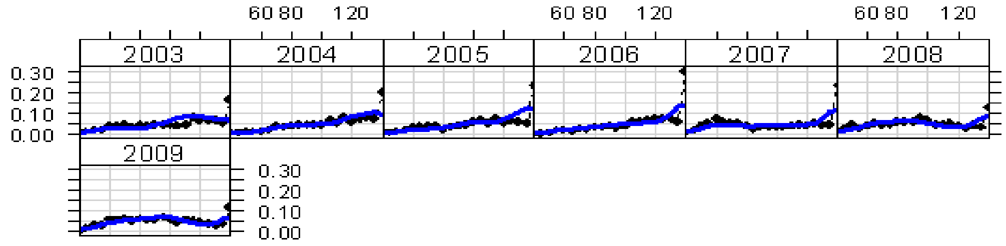


Figure B-22. NEFSC lined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (line) for Georges Bank.

(a)



(b)

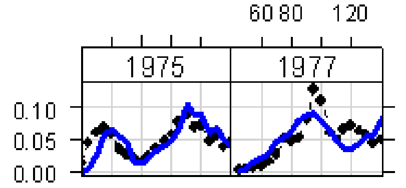


Figure B-23. Shell height proportions for (a) the SMAST large camera survey and (b) the NEFSC unlined dredge survey together with model predicted proportions (lines).

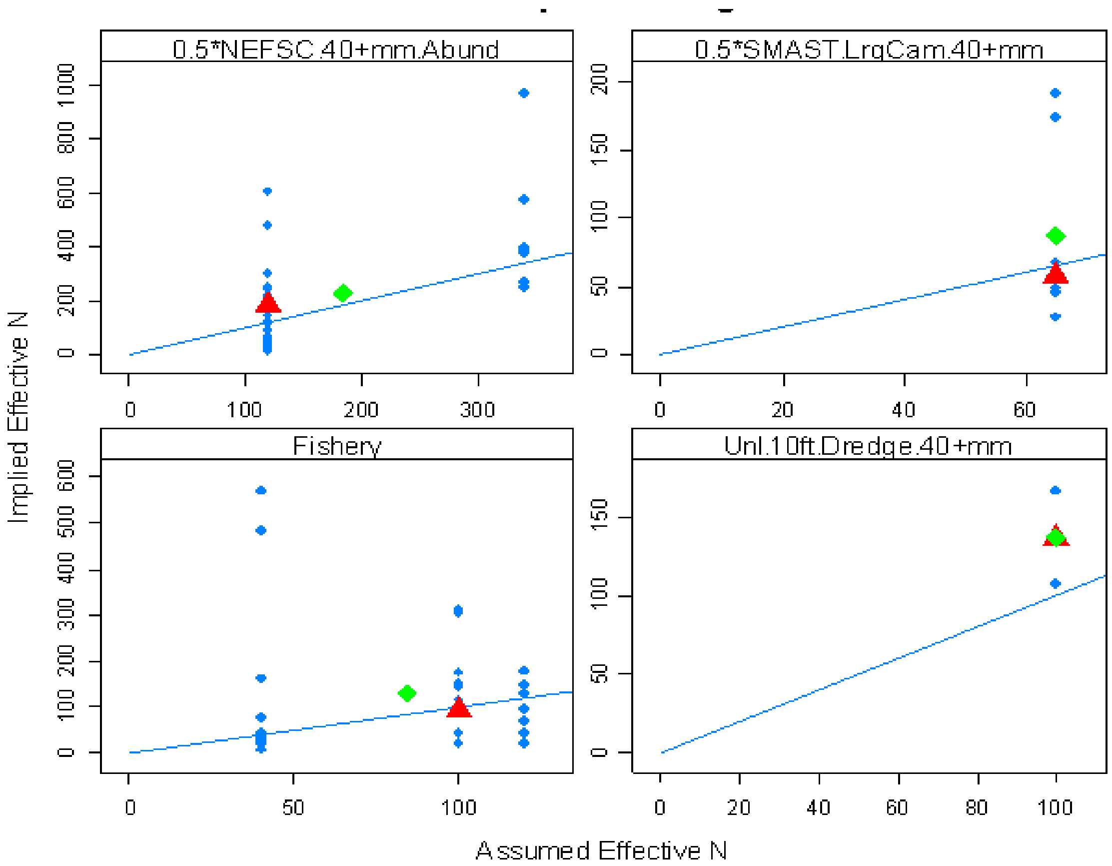


Figure B-24. Assumed and model implied effective sample sizes for the three surveys (NEFSC unlined dredge, SMAST large camera, NEFSC unlined dredge) and the fishery shell height compositions for Georges Bank.

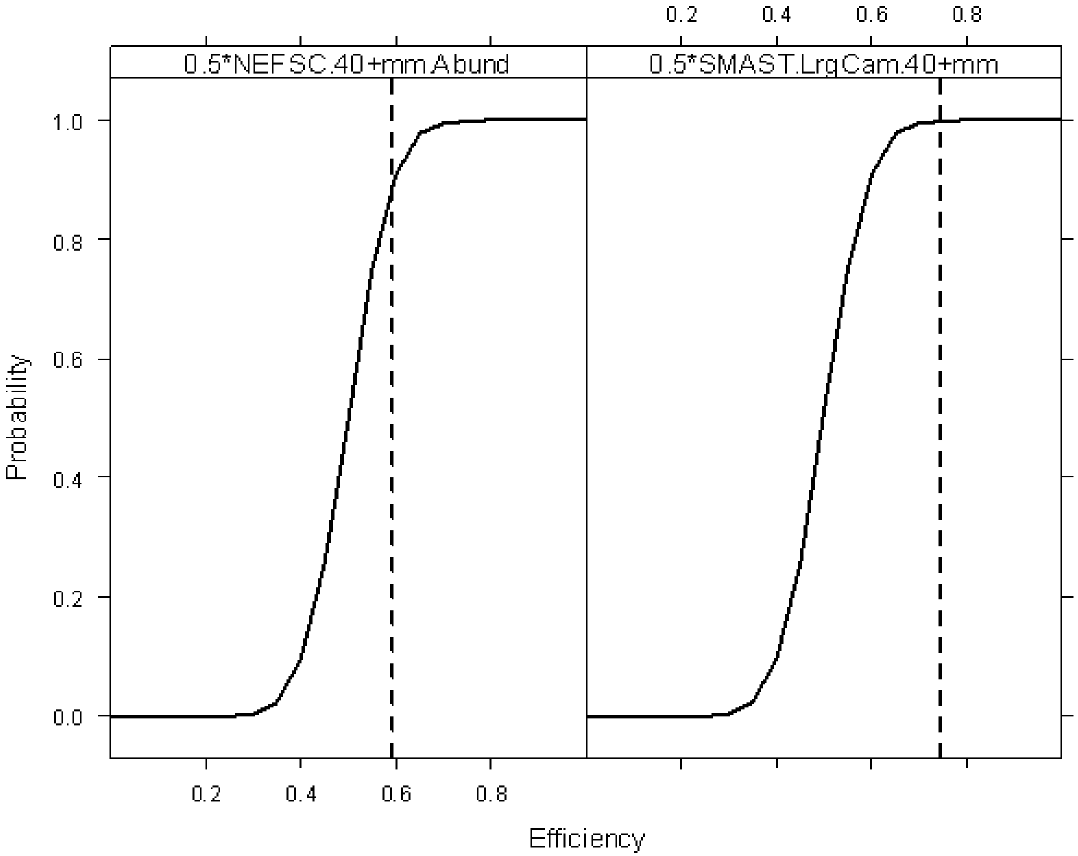
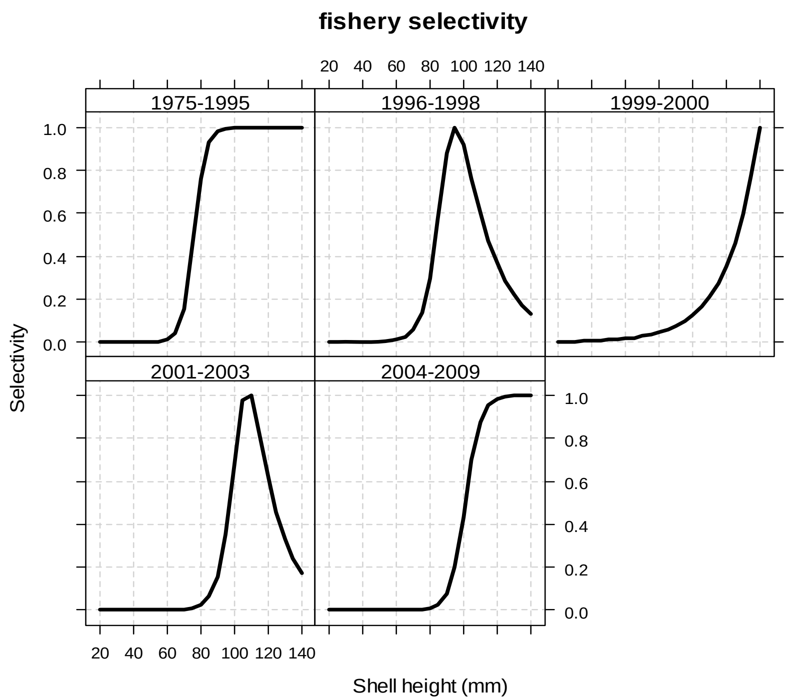


Figure B-25. Prior cumulative distributions for efficiency of the lined dredge survey (left) and large camera video survey (right) for Georges Bank. The dashed lines are the mean posterior estimate for survey efficiency.

(a)



(b)

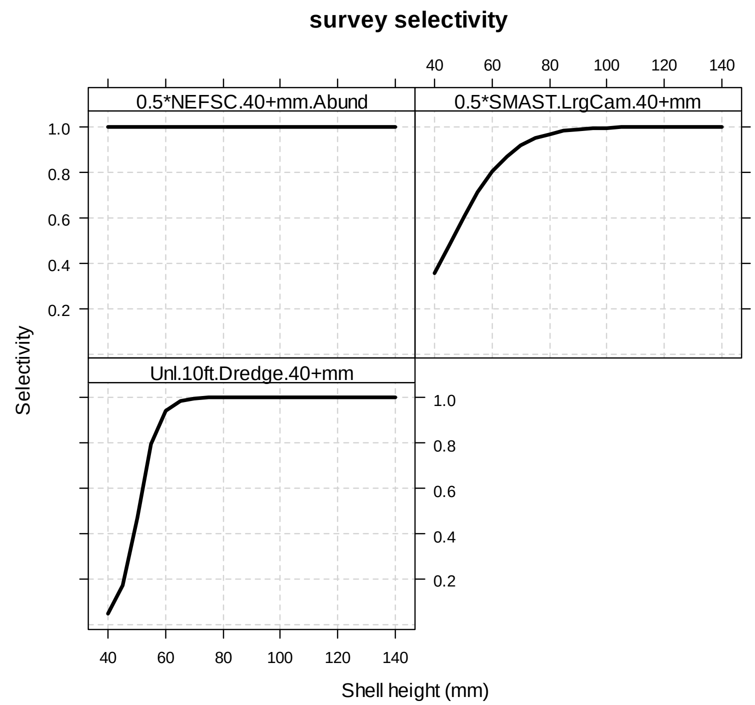


Figure B-26. (a) Estimated fishery selectivities and (b) assumed survey selectivities (lined dredge top right, large camera top left, unlined dredge bottom left) for Georges Bank.

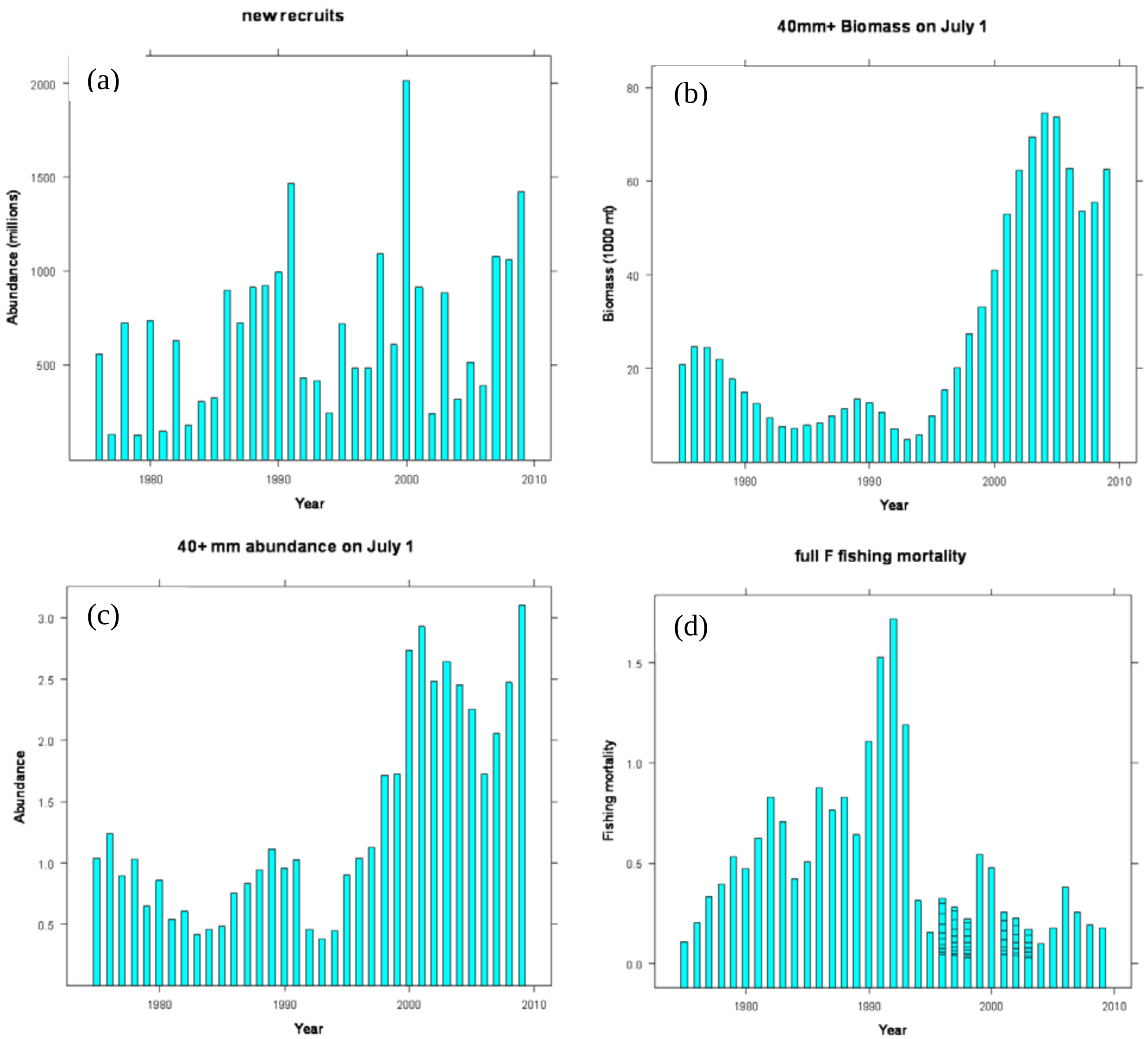


Figure B-27. CASA model estimated (a) recruitment, (b) July 1 biomass, (c) July 1 abundance and (d) fully recruited fishing mortality for Georges Bank.

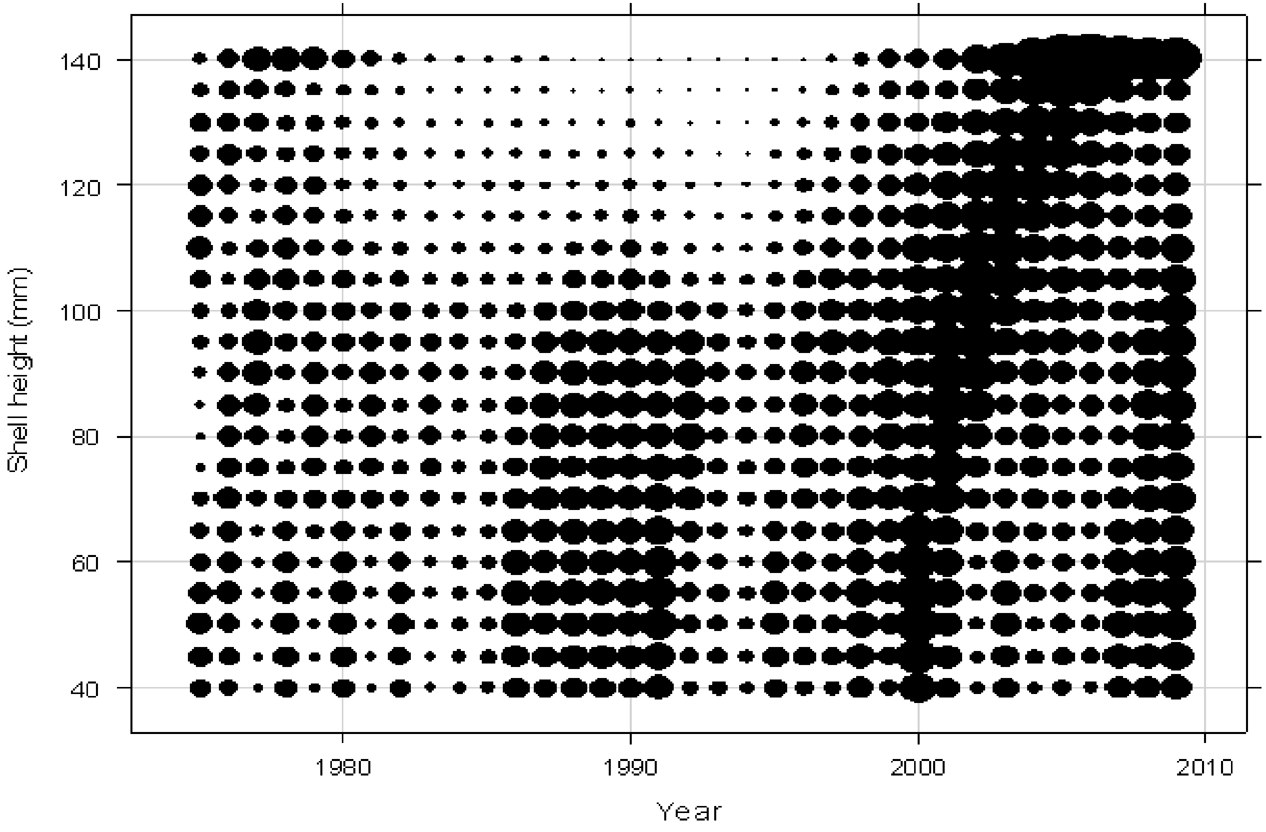


Figure B-28. Model estimated abundances at shell height for Georges Bank. Disk areas are proportional to abundance.

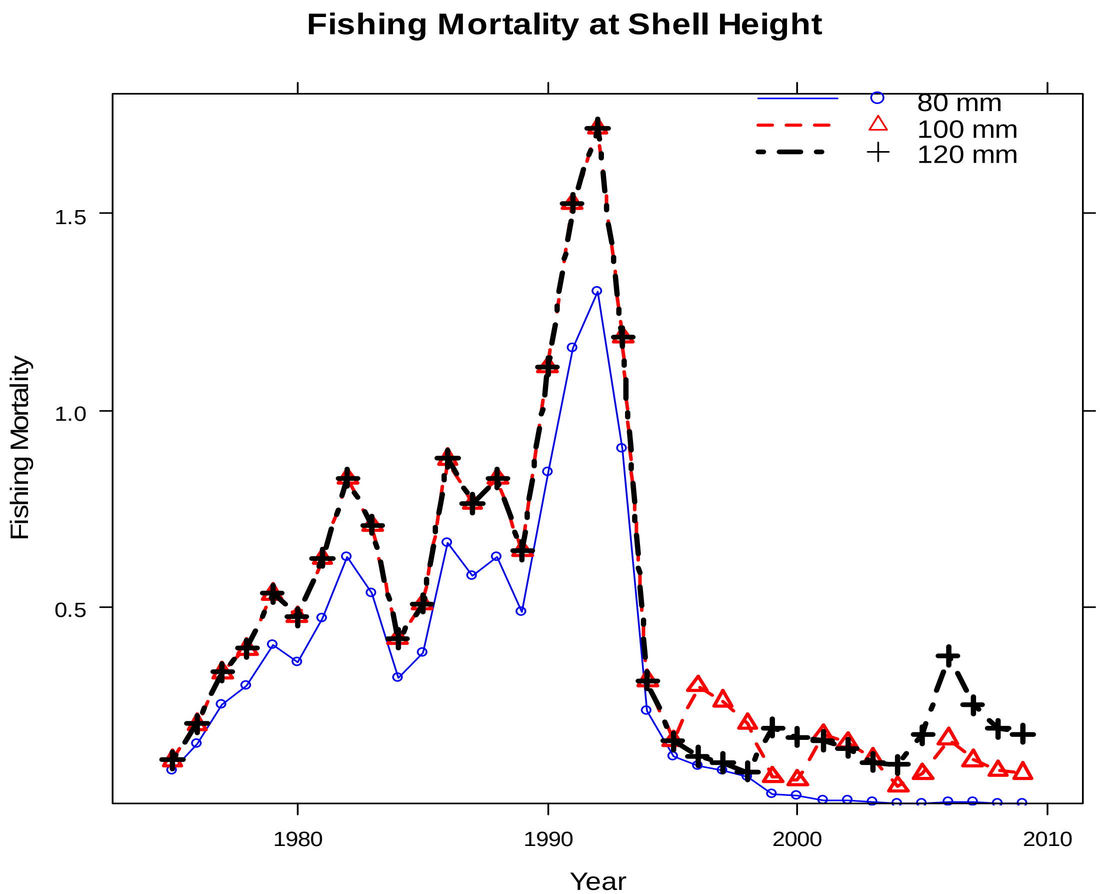
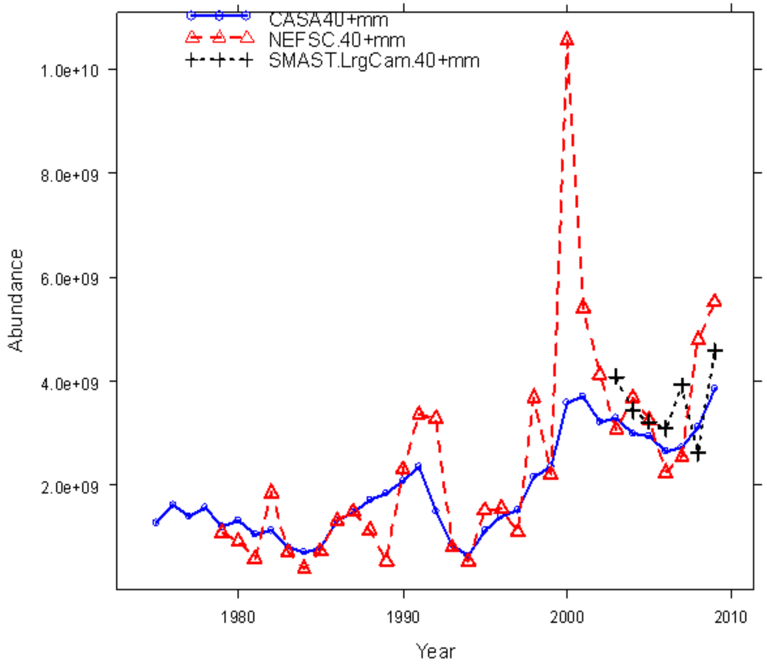


Figure B-29. CASA model estimated fishing mortality at 80 mm (blue line with circles), 100 mm (red dashed line with triangles) and 120 mm SH (black dot-dashed line with pluses) for Georges Bank.

(a)



(b)

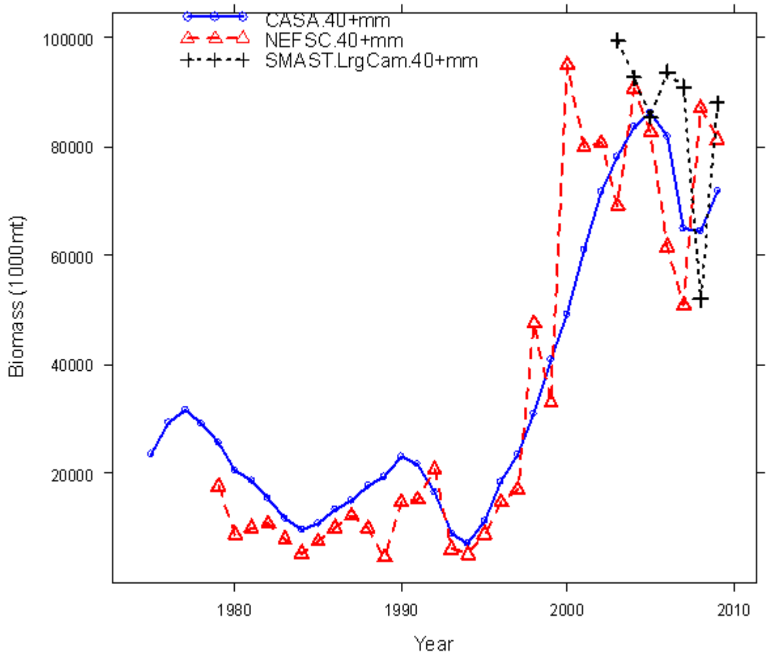


Figure B-30. Comparison of CASA model estimated (a) abundance and (b) biomass with estimates from the lined dredge survey (dashed red line with triangles) and large camera survey (dotted line with pluses) for Georges Bank. The dredge survey was expanded assuming an efficiency of 0.41.

(a)

(b)

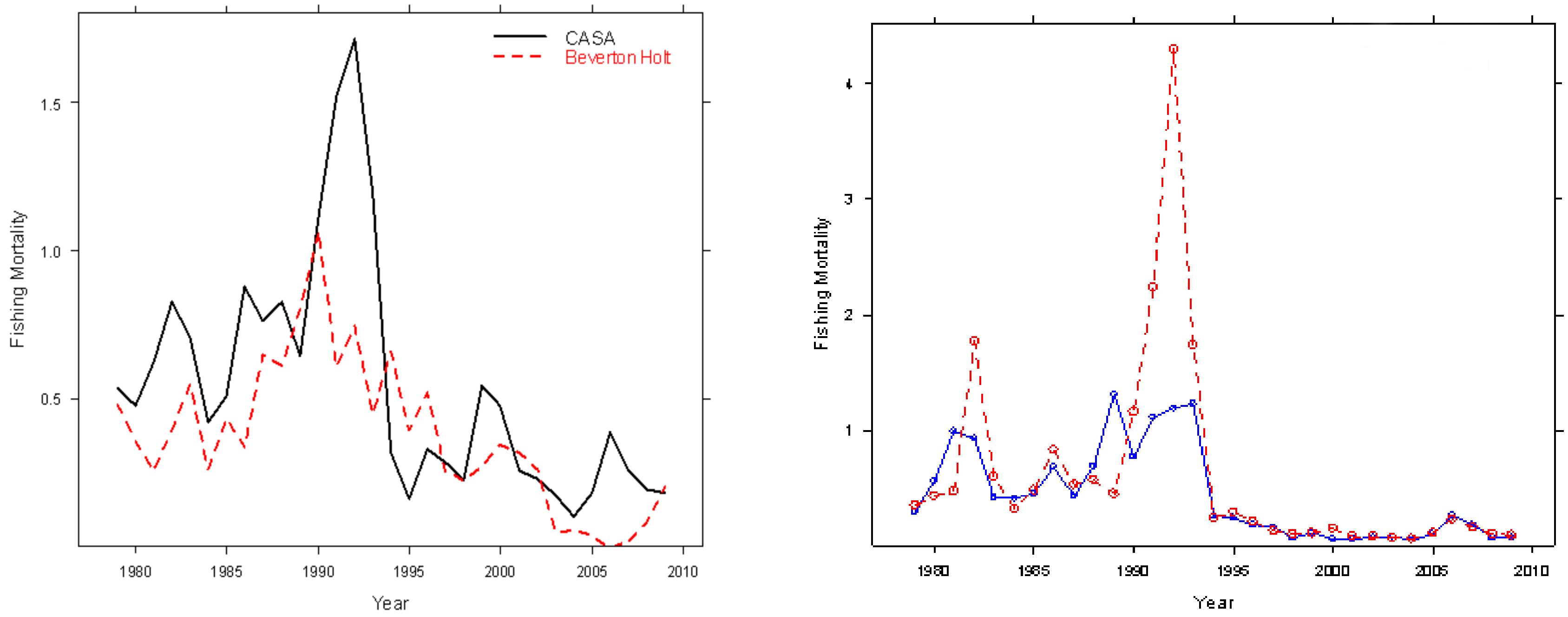


Figure B-31. (a) Comparison of fully recruited CASA fishing mortality with those calculated from the Beverton-Holt equilibrium estimator (Lc=100mm) for Georges Bank. (b) Comparison of an exploitation index (number landed/population abundance > 80mm) based on the fishery and lined dredge survey data (red dotted line), and CASA model (blue solid line) for Georges Bank.

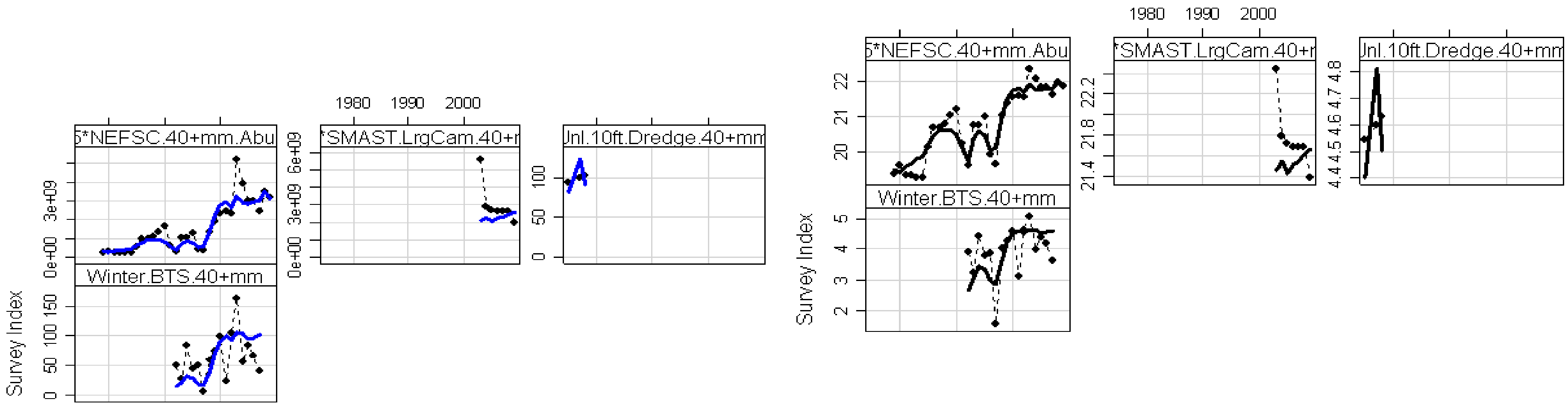


Figure B-32. Comparison between survey trend data (solid circles) and corresponding model estimates (lines) for the NEFSC lined dredge survey, the SMAST large camera survey, the NEFSC unlined dredge and winter trawl surveys for the Mid-Atlantic. Results are shown on a linear scale (left) and a log scale (right).

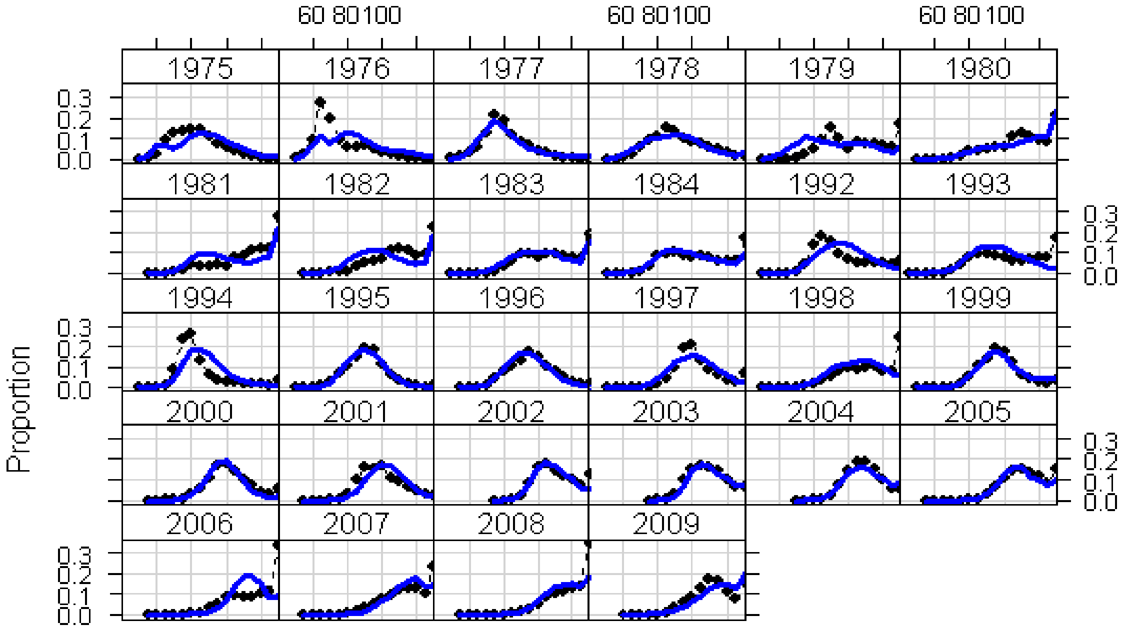


Figure B-33. Comparison of fishery shell height proportions (solid circles) and model estimated fishery shell height proportions (lines) for the Mid-Atlantic.

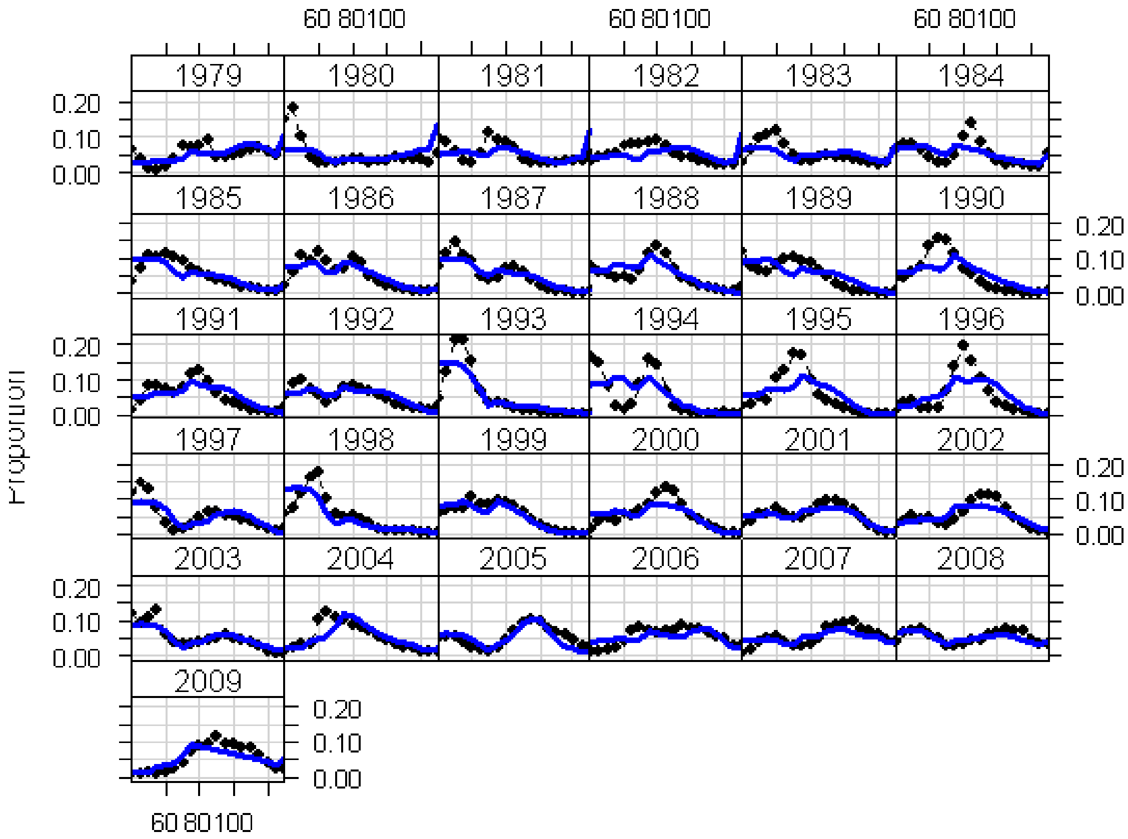
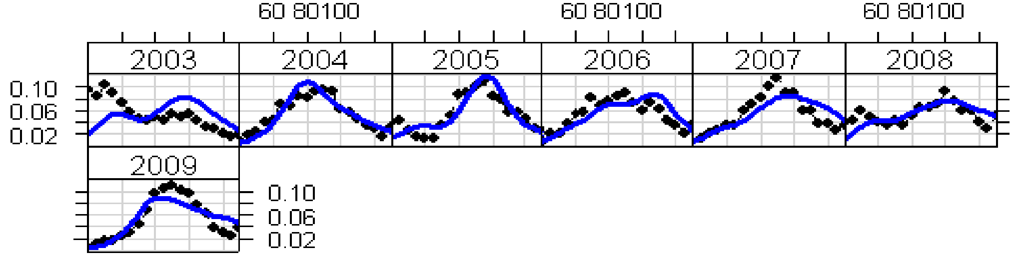
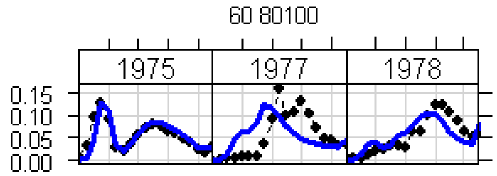


Figure B-34. NEFSC lined dredge survey shell height proportions (solid circles) and model estimated shell height proportions (line) for the Mid-Atlantic.

(a)



(b)



(c)

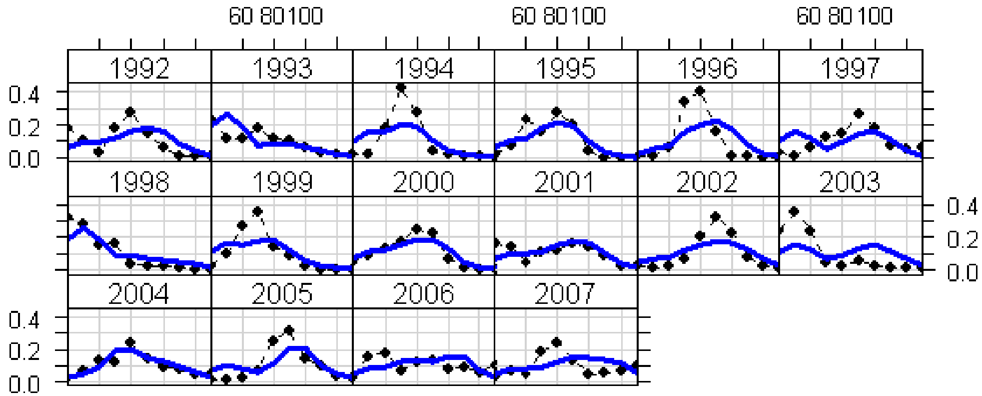


Figure B-35. Shell height proportions for (a) the SMAST large camera survey (b) the NEFSC unlined dredge survey and (c) the NEFSC winter trawl survey together with model predicted proportions (lines) in the Mid-Atlantic.

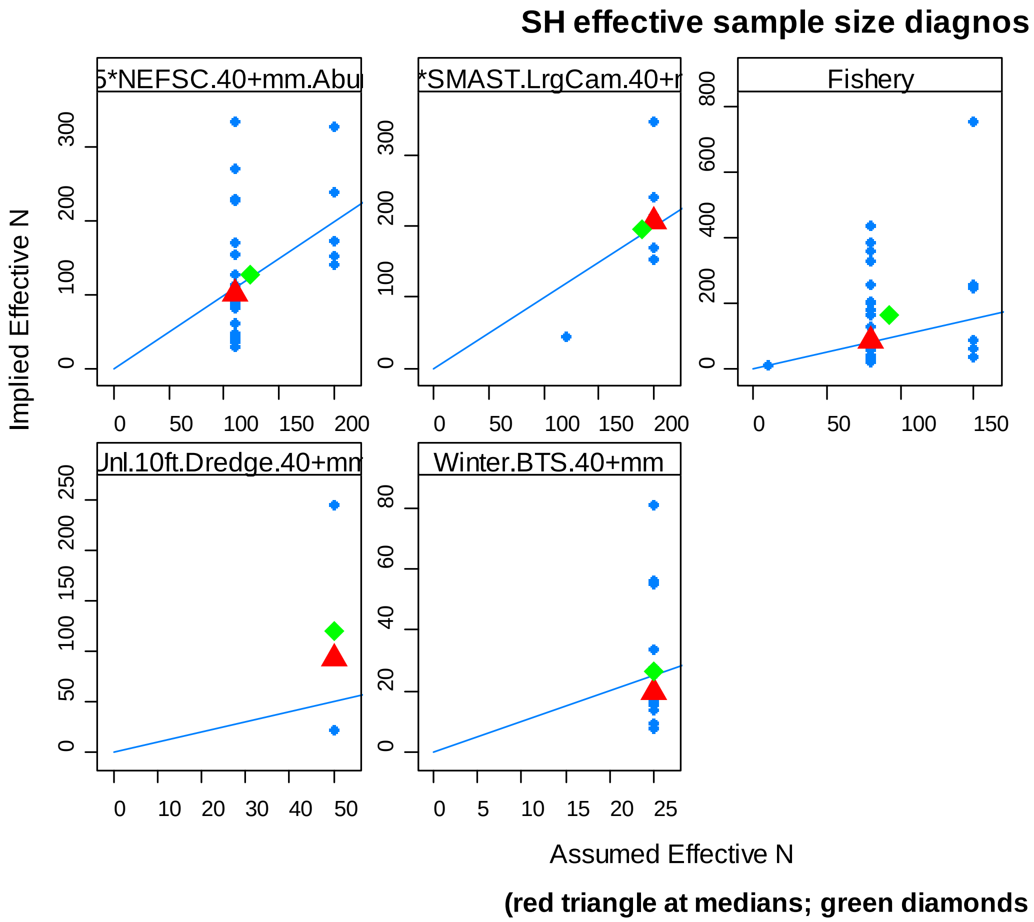


Figure B-36. Assumed and model implied effective sample sizes for the four surveys (NEFSC unlined dredge, SMAST large camera, NEFSC unlined dredge, winter bottom trawl) and the fishery shell height compositions for the Mid-Atlantic.

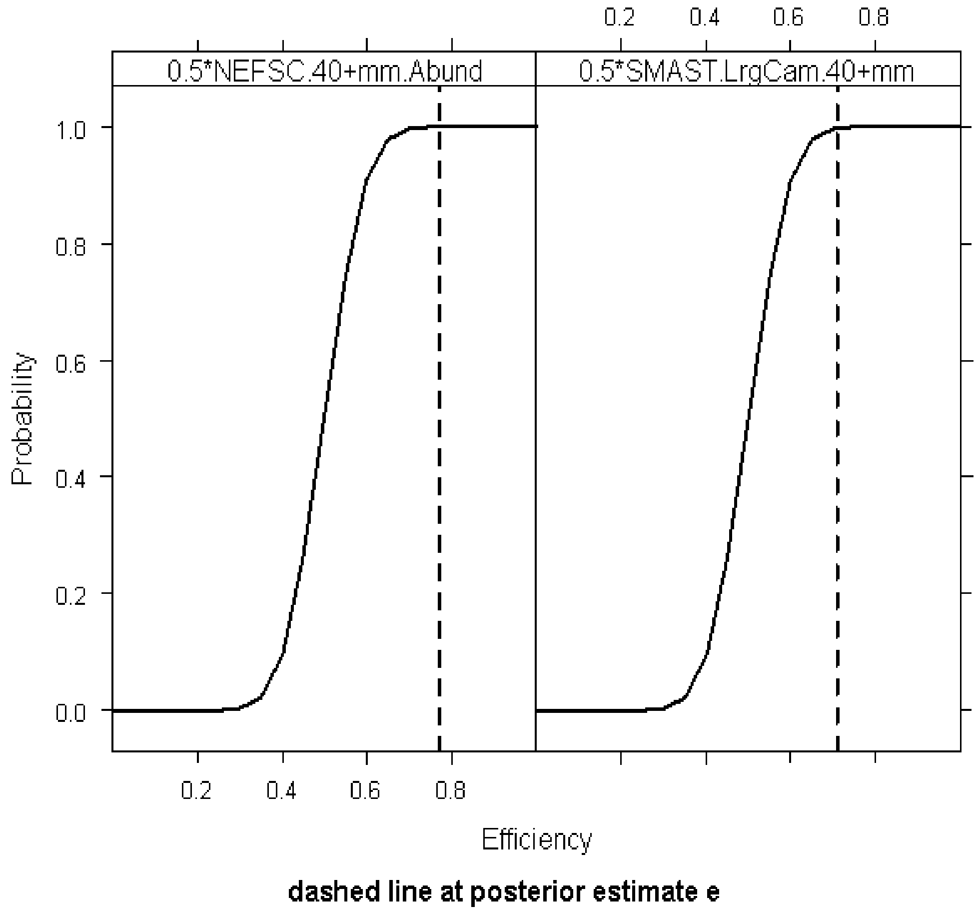


Figure B-37. Prior cumulative distributions for efficiency of the lined dredge survey (left) and large camera video survey (right) for the Mid-Atlantic. The dashed lines are the mean posterior estimate for survey efficiency.

(a) (b)

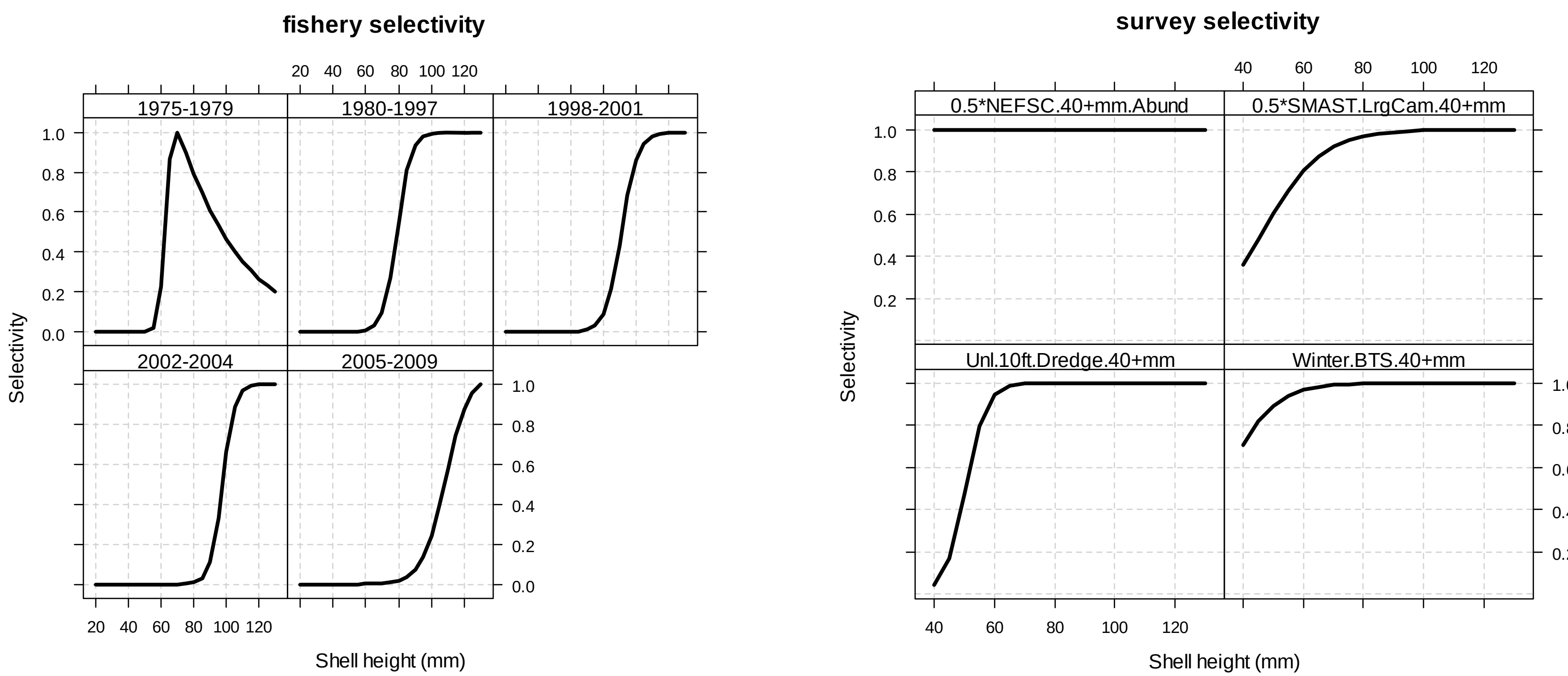


Figure B-38. (a) Estimated fishery selectivities and (b) survey selectivities (lined dredge top left, large camera top right, unlined dredge bottom left, winter trawl bottom right) for the Mid-Atlantic. The trawl survey selectivity was estimated; the other survey selectivities were fixed.

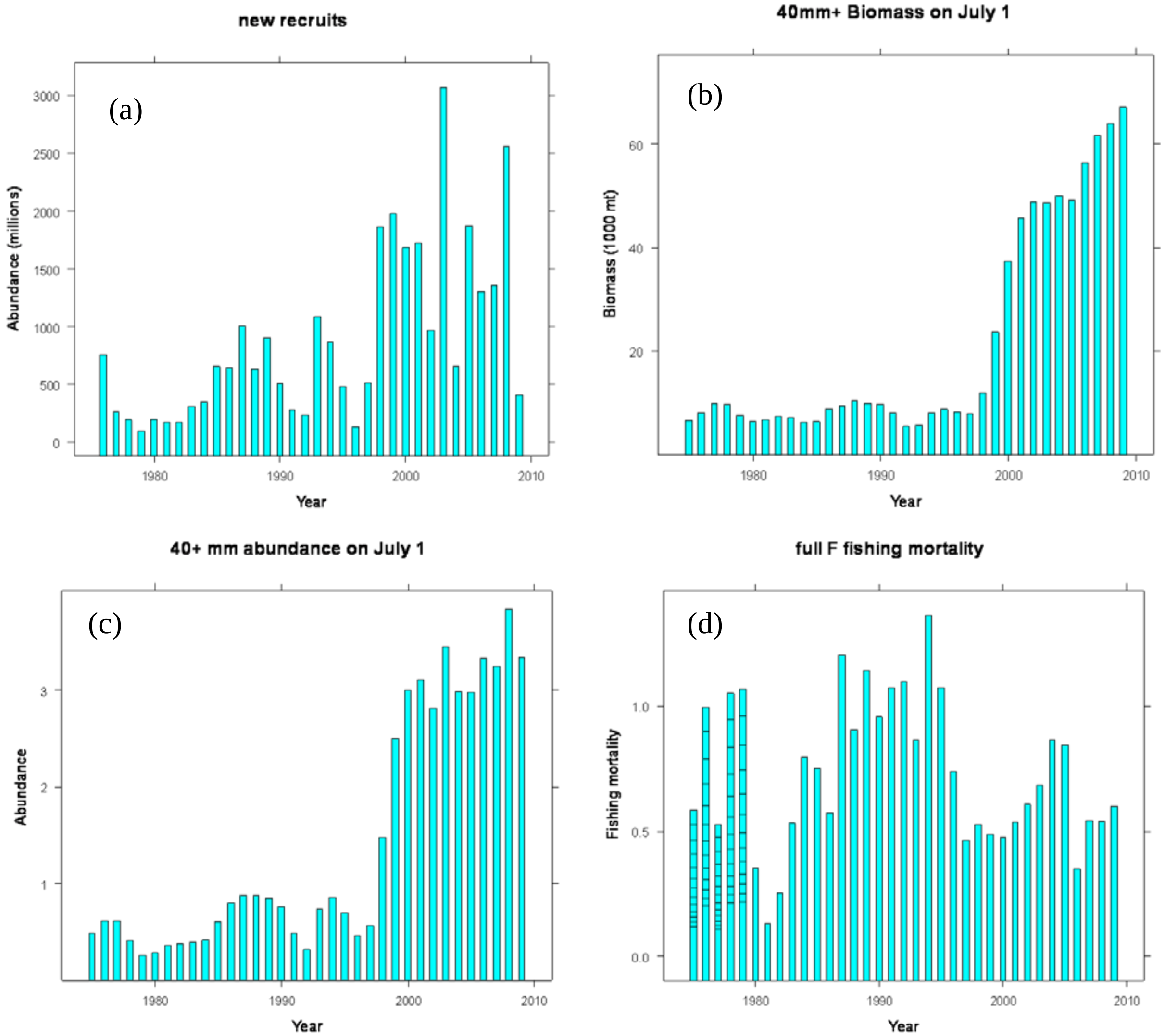


Figure B-39. CASA model estimated (a) recruitment, (b) July 1 biomass, (c) July 1 abundance and (d) fully recruited fishing mortality for the Mid-Atlantic.

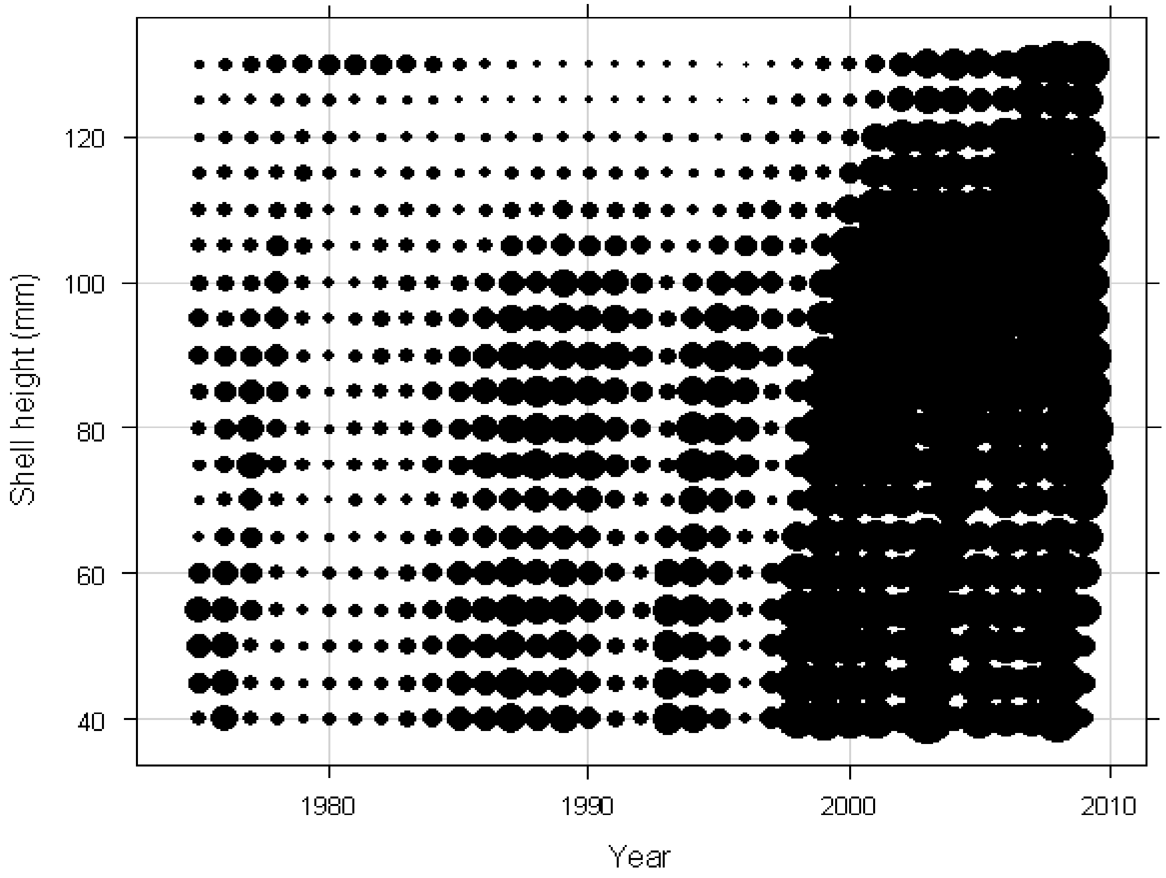


Figure B-40. Model estimated abundances at shell height for the Mid-Atlantic. Disk areas are proportional to abundance.

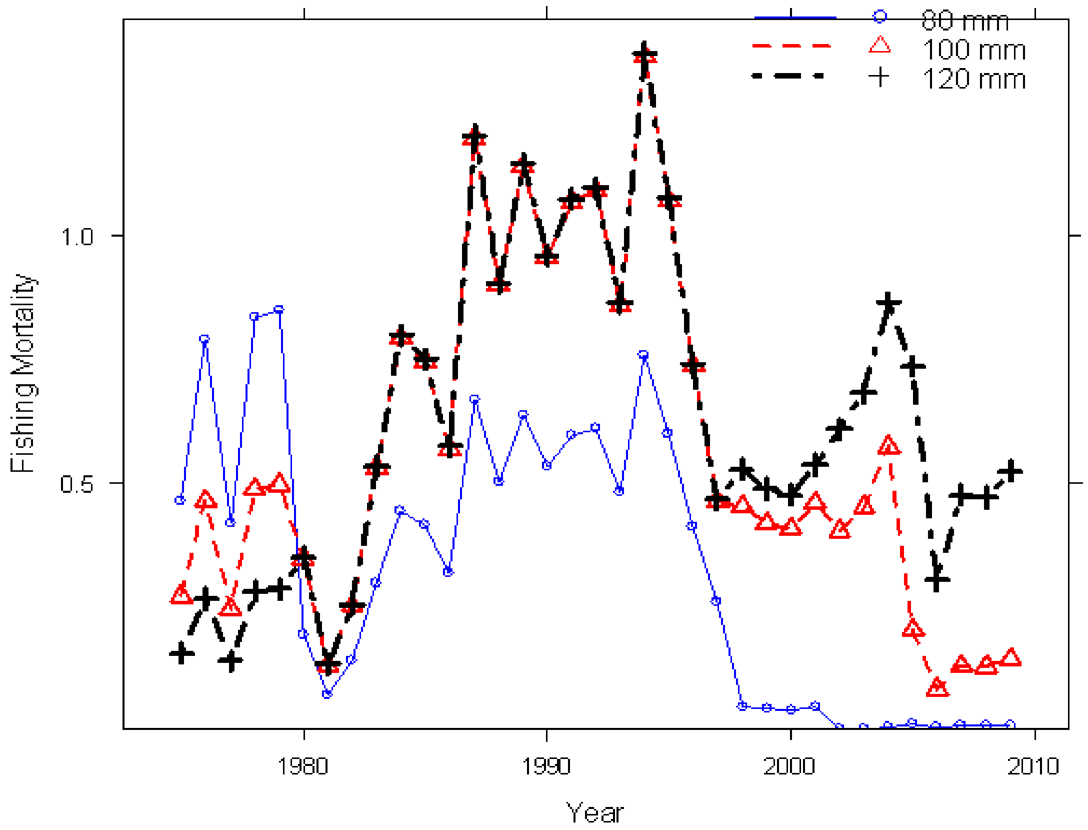


Figure B-41. CASA model estimated fishing mortality at 80 mm (blue line with circles), 100 mm (red dashed line with triangles) and 120 mm SH (black dot-dashed line with pluses) for the Mid-Atlantic.

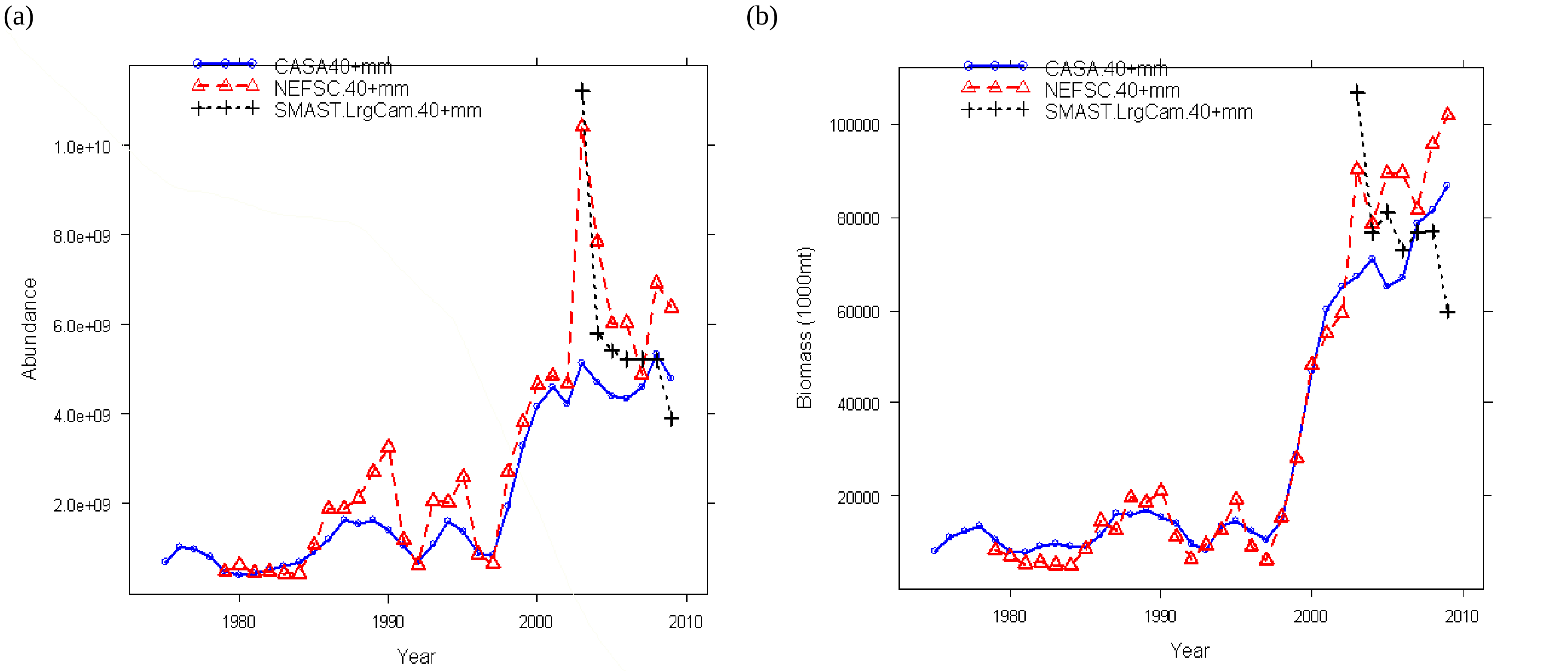
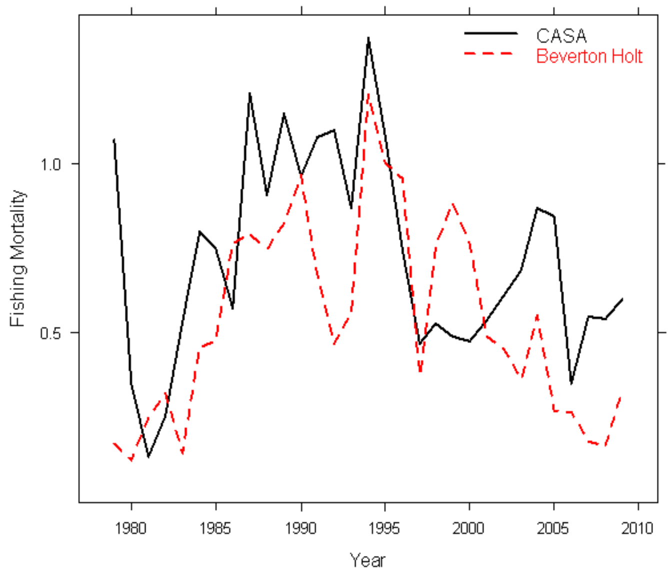


Figure B-42. Comparison of CASA model estimated (a) abundance and (b) biomass with estimates from the lined dredge survey (dashed red line with triangles) and large camera survey (dotted line with pluses) for the Mid-Atlantic. The dredge survey was expanded assuming an efficiency of 0.44.

(a)



(b)

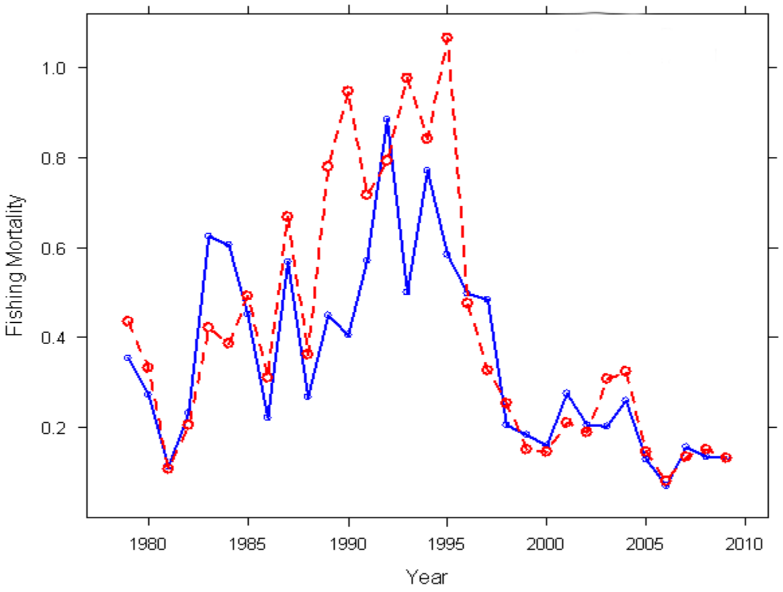


Figure B-43. (a) Comparison of fully recruited CASA fishing mortality with those calculated from the Beverton-Holt equilibrium estimator (Lc=100mm) for the Mid-Atlantic. (b) Comparison of an exploitation index (number landed/population abundance > 80mm) based on the fishery and lined dredge survey data (red dashed line), and CASA model (blue solid line) for the Mid-Atlantic.

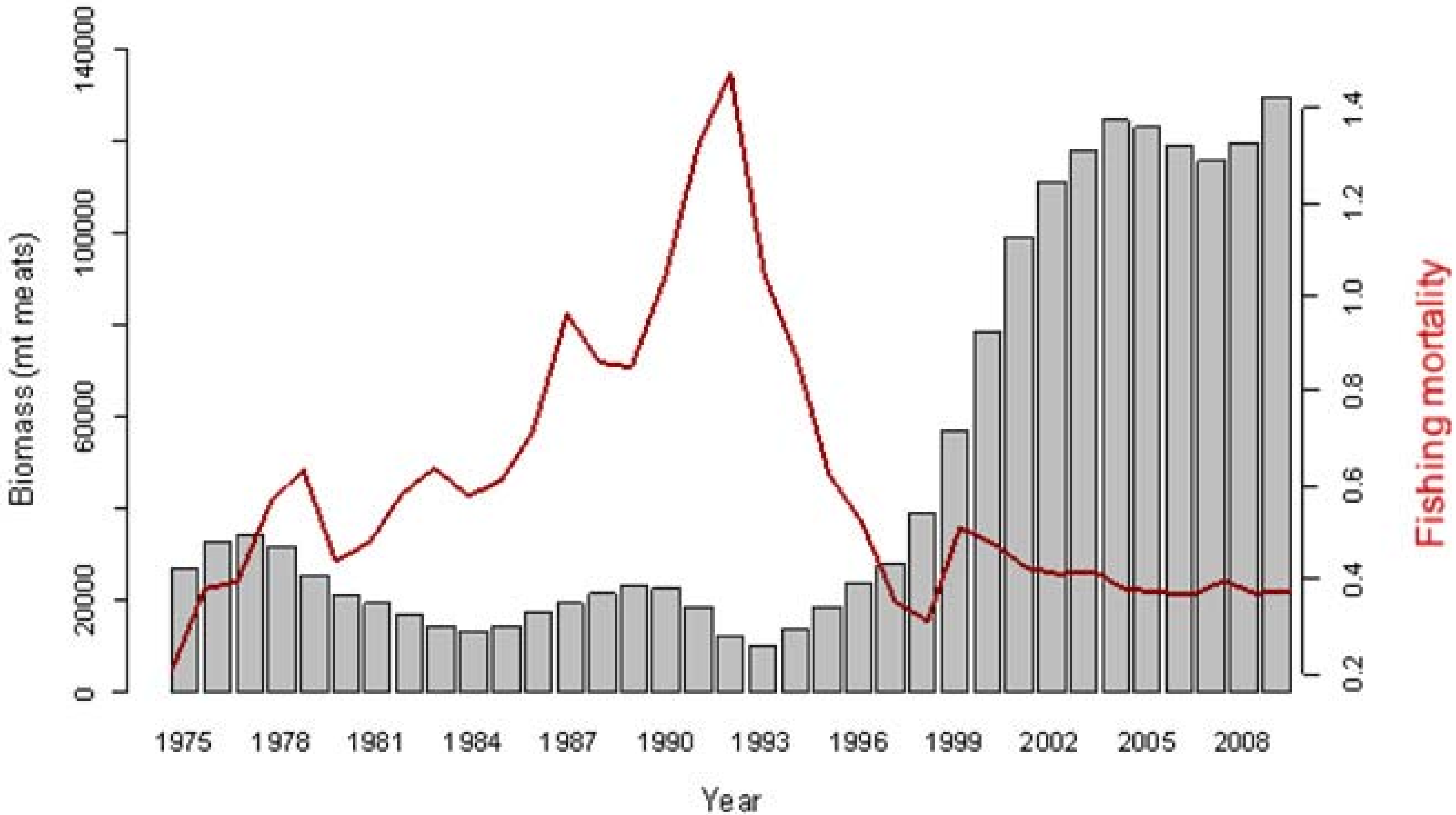


Figure B-44. Whole-stock CASA model estimates of biomass (bars) and fully recruited fishing mortality (line).

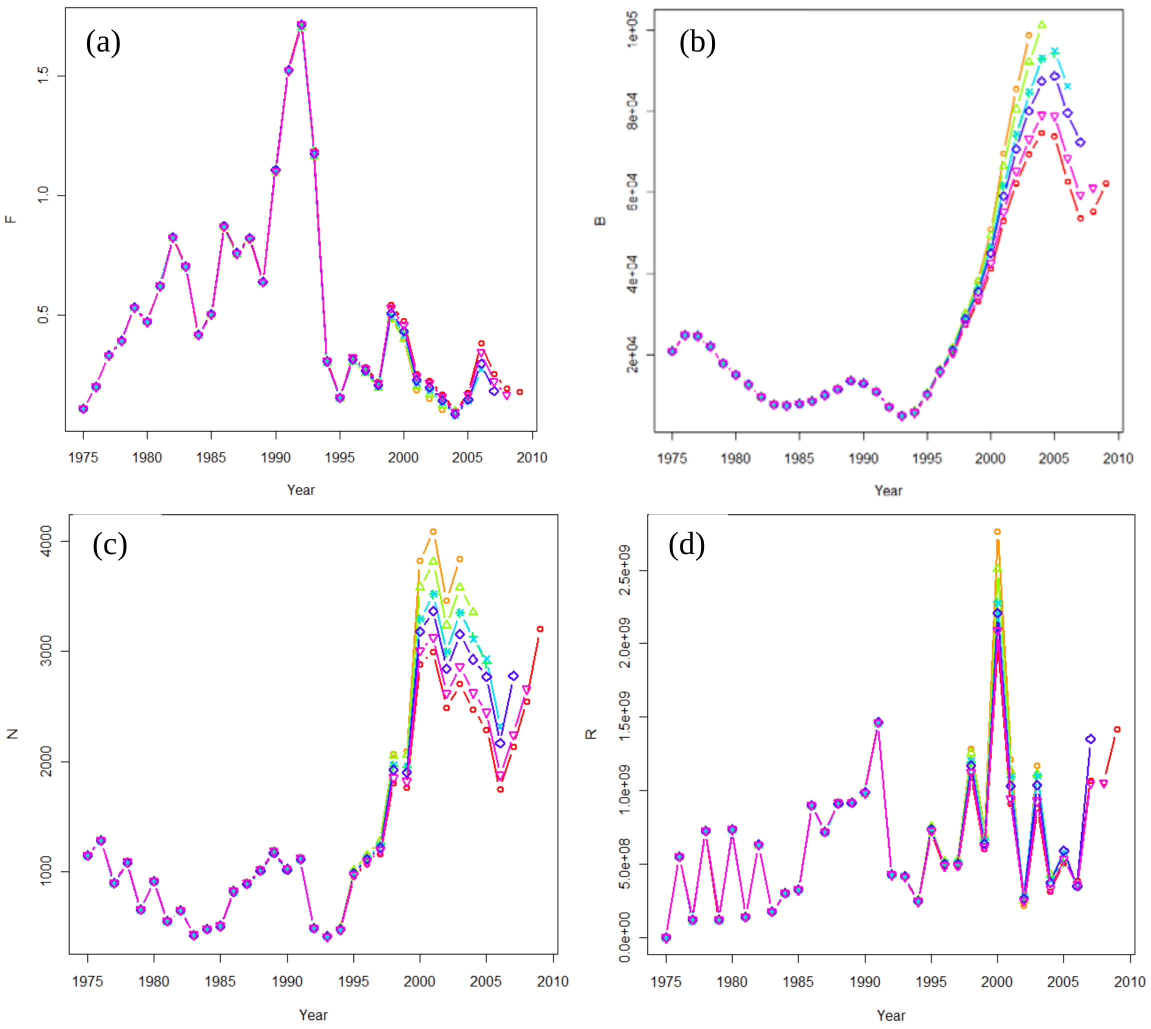


Figure B-45. Plots of retrospective analysis of the Georges Bank CASA model: (a) fishing mortality, (b) biomass, (c) abundance, and (d) recruitment. The CASA model was run with terminal year 2004 (orange), 2005 (green), 2006 (cyan), 2007 (blue), 2008 (magenta) and 2009 (red).

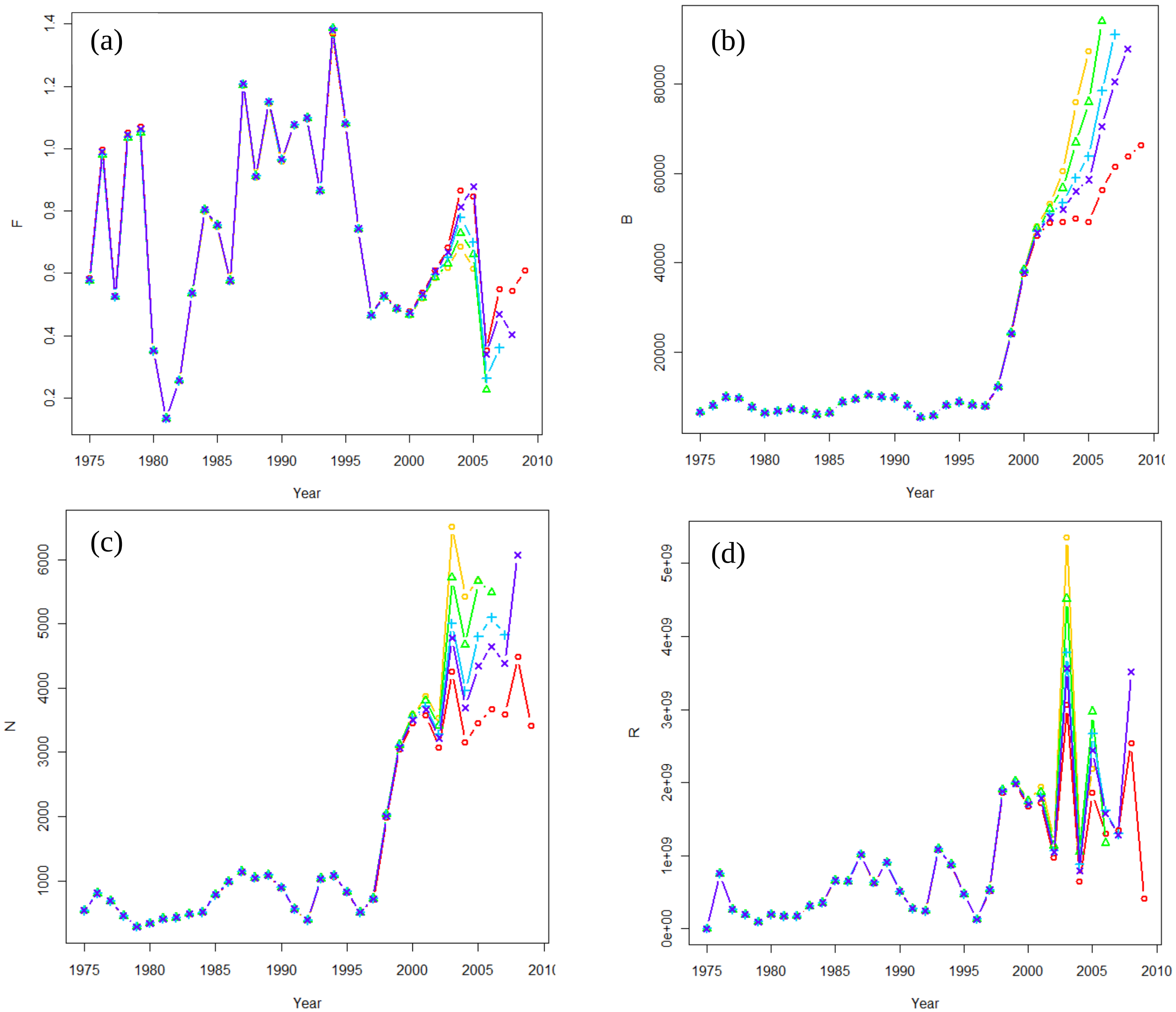


Figure B-46. Plots of retrospective analysis of the Mid-Atlantic CASA model: (a) fishing mortality, (b) biomass, (c) abundance, and (d) recruitment. The CASA model was run with terminal years 2004 (orange), 2005 (green), 2006 (cyan), 2007 (blue), 2008 (purple) and 2009 (red).

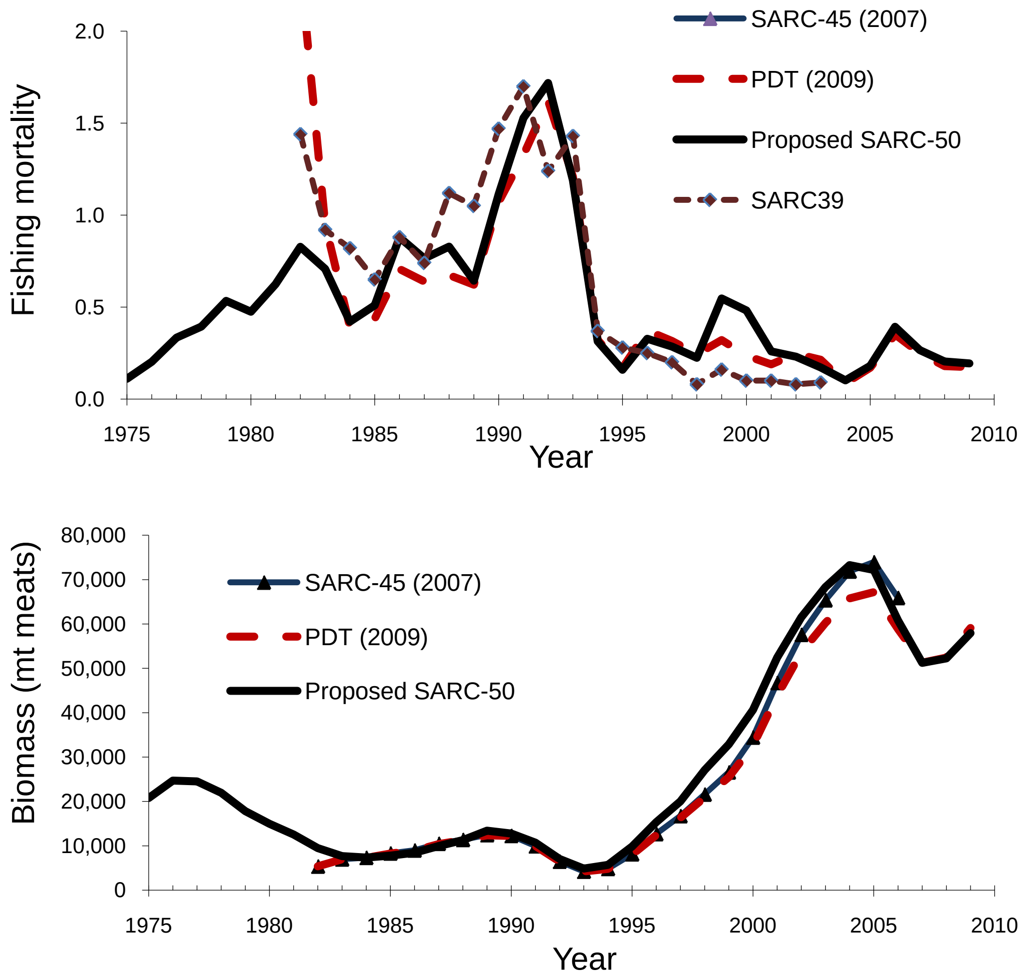


Figure B-47. Comparison of current estimates (black line) of fully recruited fishing mortality (above) and July 1 biomass (below) on Georges Bank with that of previous assessments (SARC-39/NEFSC 2004 short dashed line (fishing mortality only), SARC-45/NEFSC 2007, blue line with triangles, update assessment by the scallop PDT in 2009 (NEFMC 2010), long red dashed line).

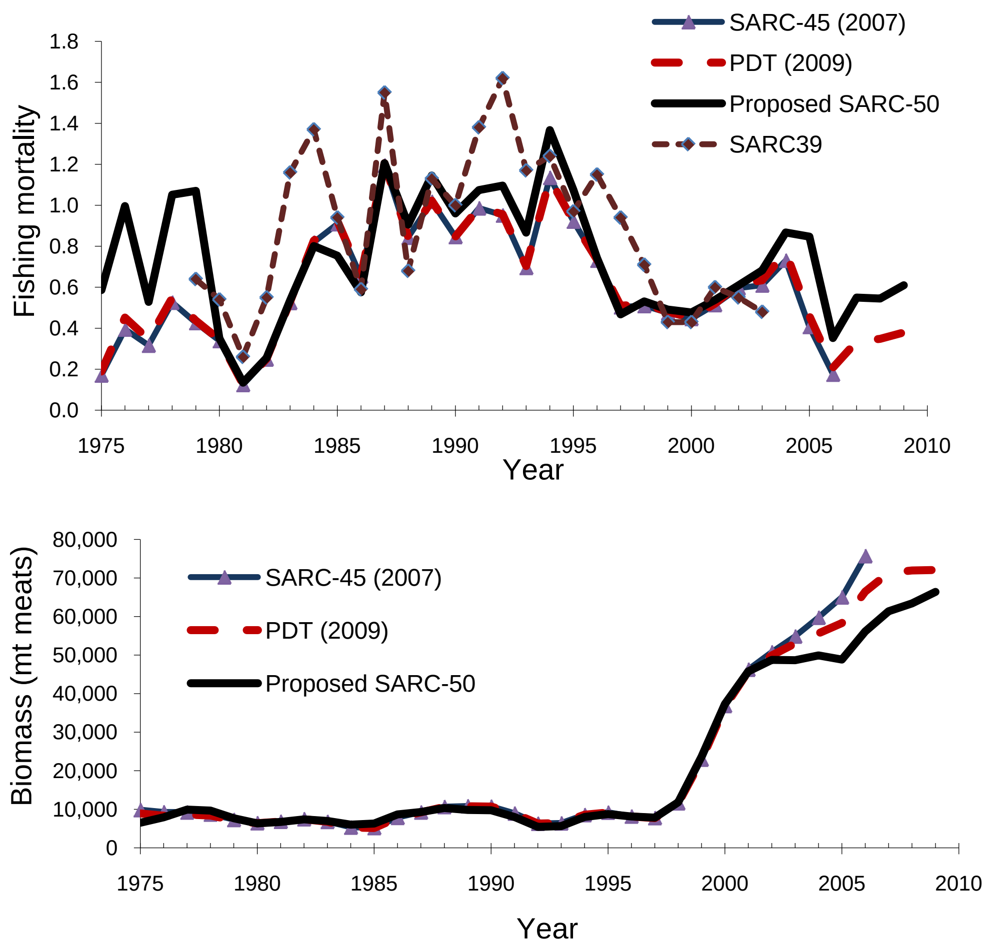


Figure B-48. Comparison of current estimates (black line) of fishing mortality (above) and biomass (below) in the Mid-Atlantic with that of previous assessments (SARC-39/NEFSC 2004 short dashed line (fishing mortality only), SARC-45/NEFSC 2007, blue line with triangles, update assessment by the scallop PDT in 2009 (NEFMC 2010), long red dashed line).

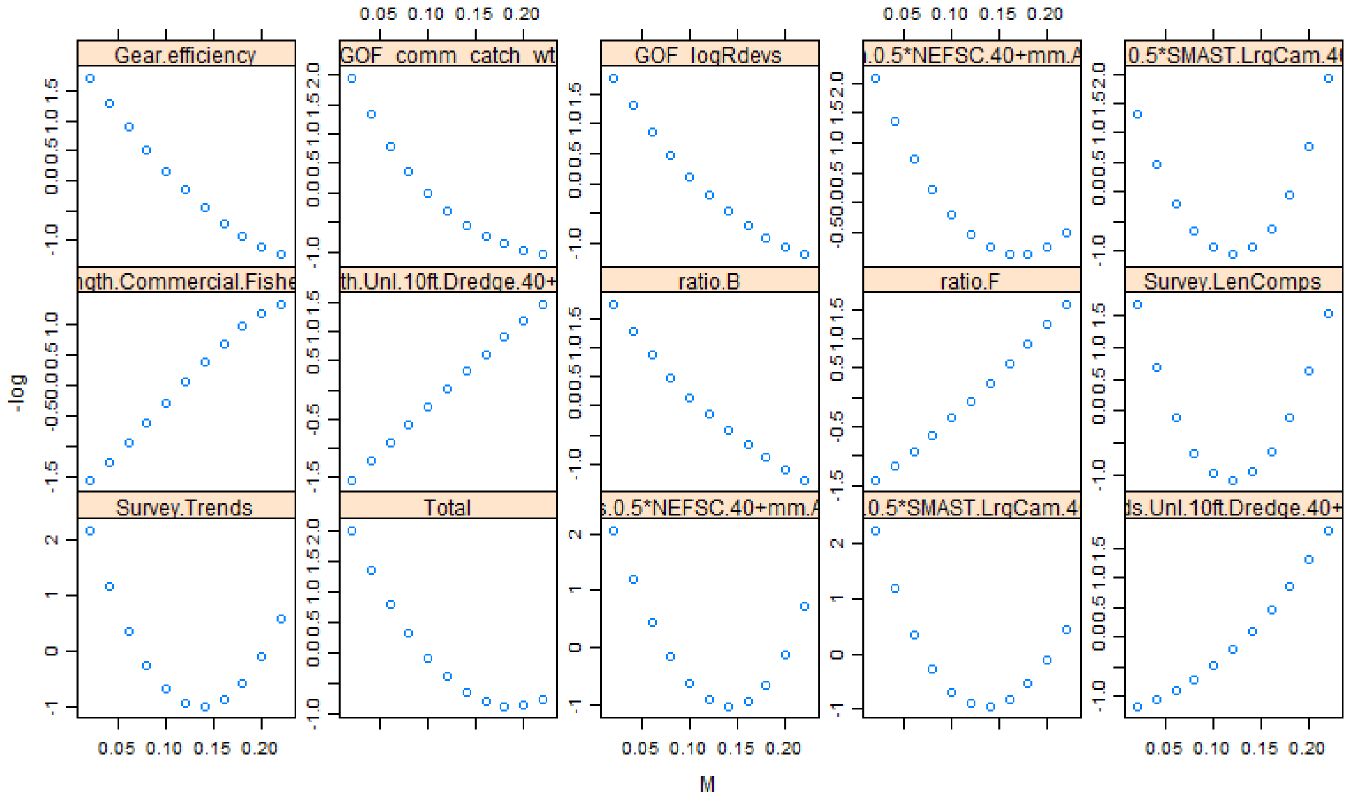
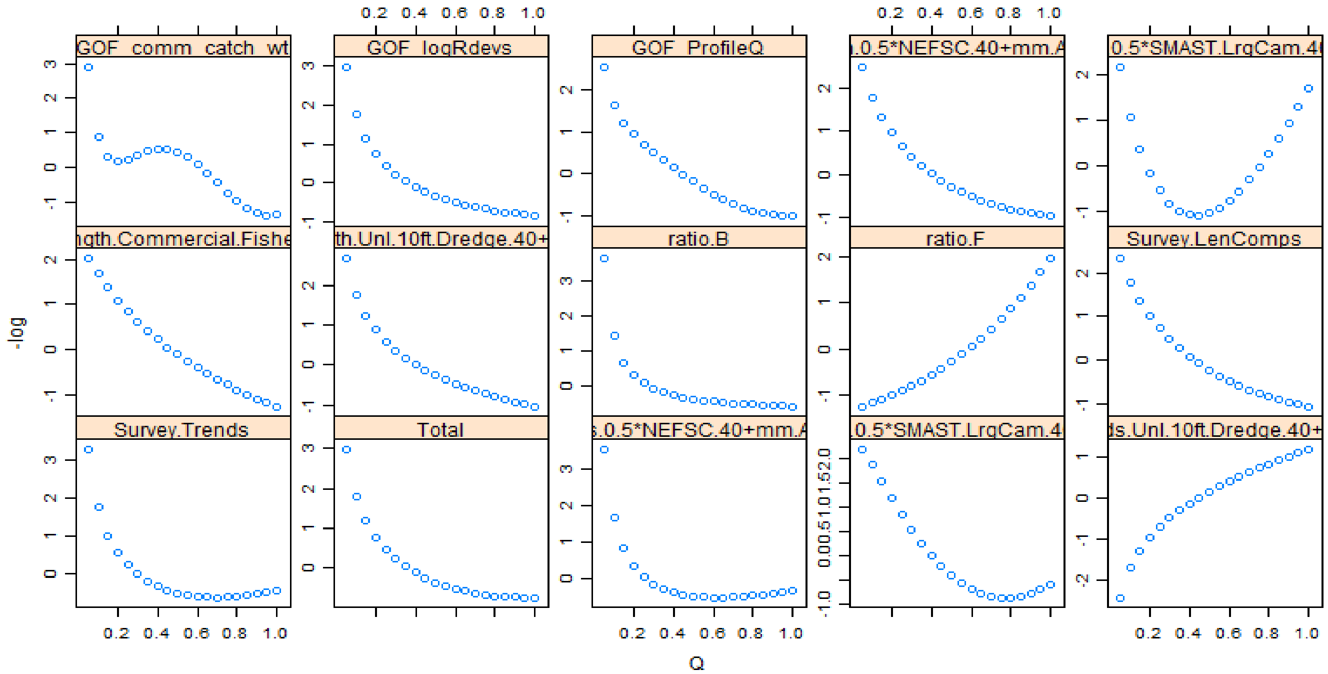


Figure B-49 Likelihood profile for (a) natural mortality and (b) large camera efficiency (*q*) on Georges Bank.



GOF_comm_catch_wt
GOF_logRdevs
GOF_ProfileQ
.0.5*NEFSC.40+mm.A
0.5*SMAST.LrgCam.40
ngth.Commercial.Fishe
th.Unl.10ft.Dredge.40+
ratio.B
ratio.F
Survey.LenComps
Survey.Trends
Total
s.0.5*NEFSC.40+mm.A
0.5*SMAST.LrgCam.4
ds.Unl.10ft.Dredge.40+
-log
Q

Figure B-49(b)

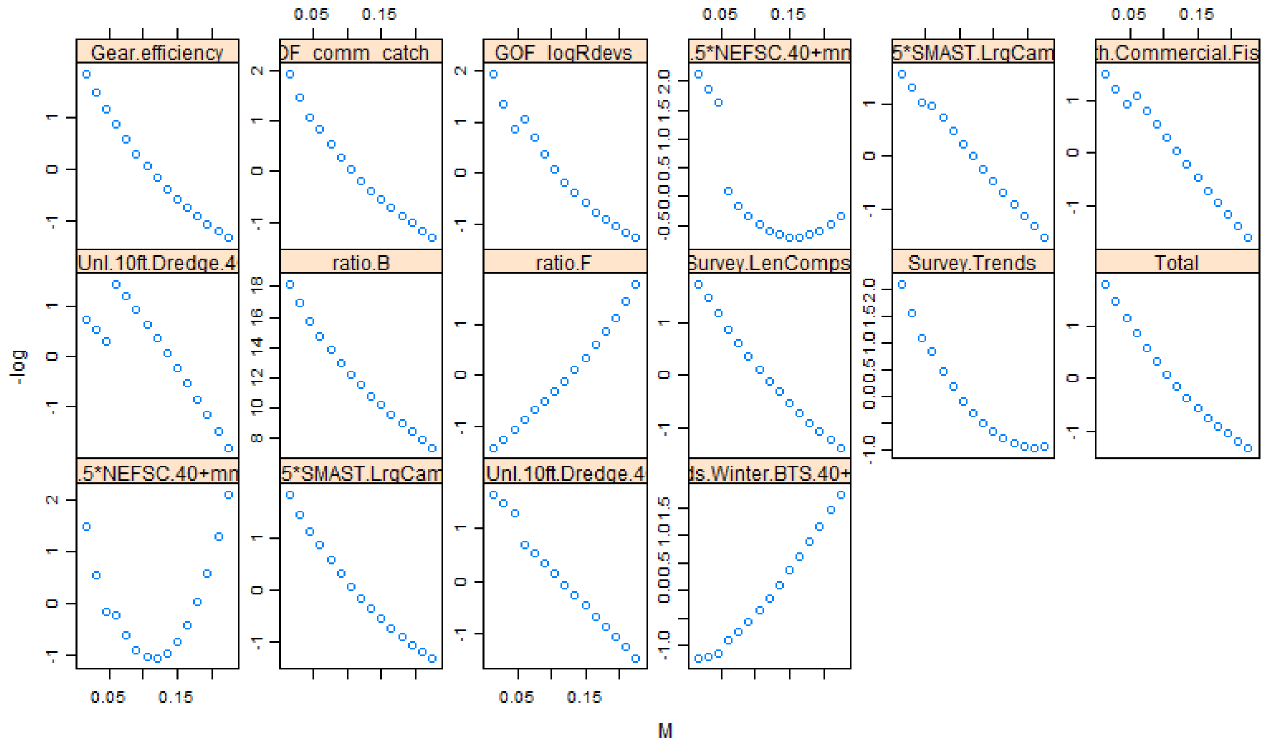


Figure B-50. Likelihood profiles for (a) natural mortality and (b) large camera survey *q* for the Mid-Atlantic.

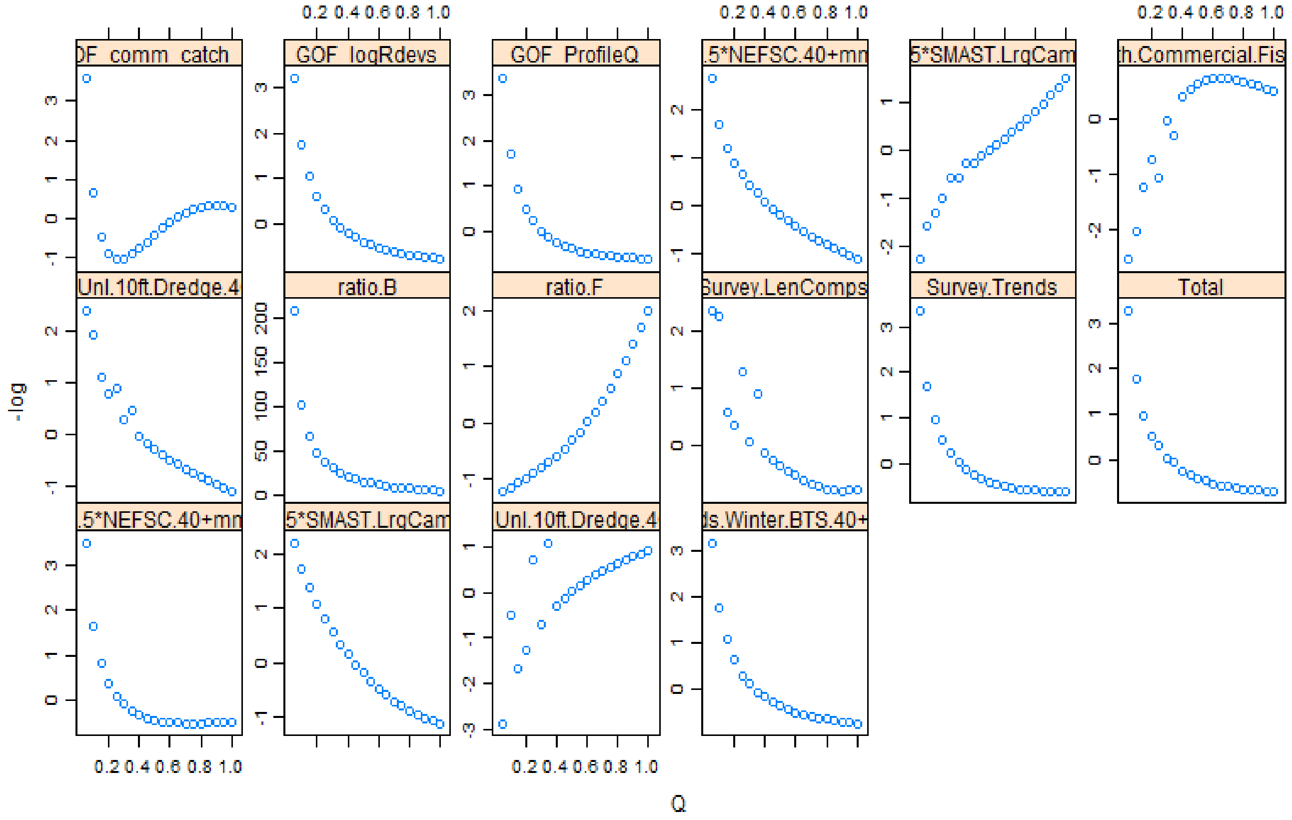

0.2 0.4 0.6 0.8 1.0
OF_comm_catch
GOF_logRdevs
GOF_ProfileQ
.5*NEFSC.40+mn
5*SMAST.LrgCam
h.Commercial.Fis
Unl.10ft.Dredge.4
ratio.B
ratio.F
Survey.LenComps
Survey.Trends
Total
.5*NEFSC.40+mn
5*SMAST.LrgCam
Unl.10ft.Dredge.4
ls.Winter.BTS.40+
-log
Q

Figure B-50 (b)

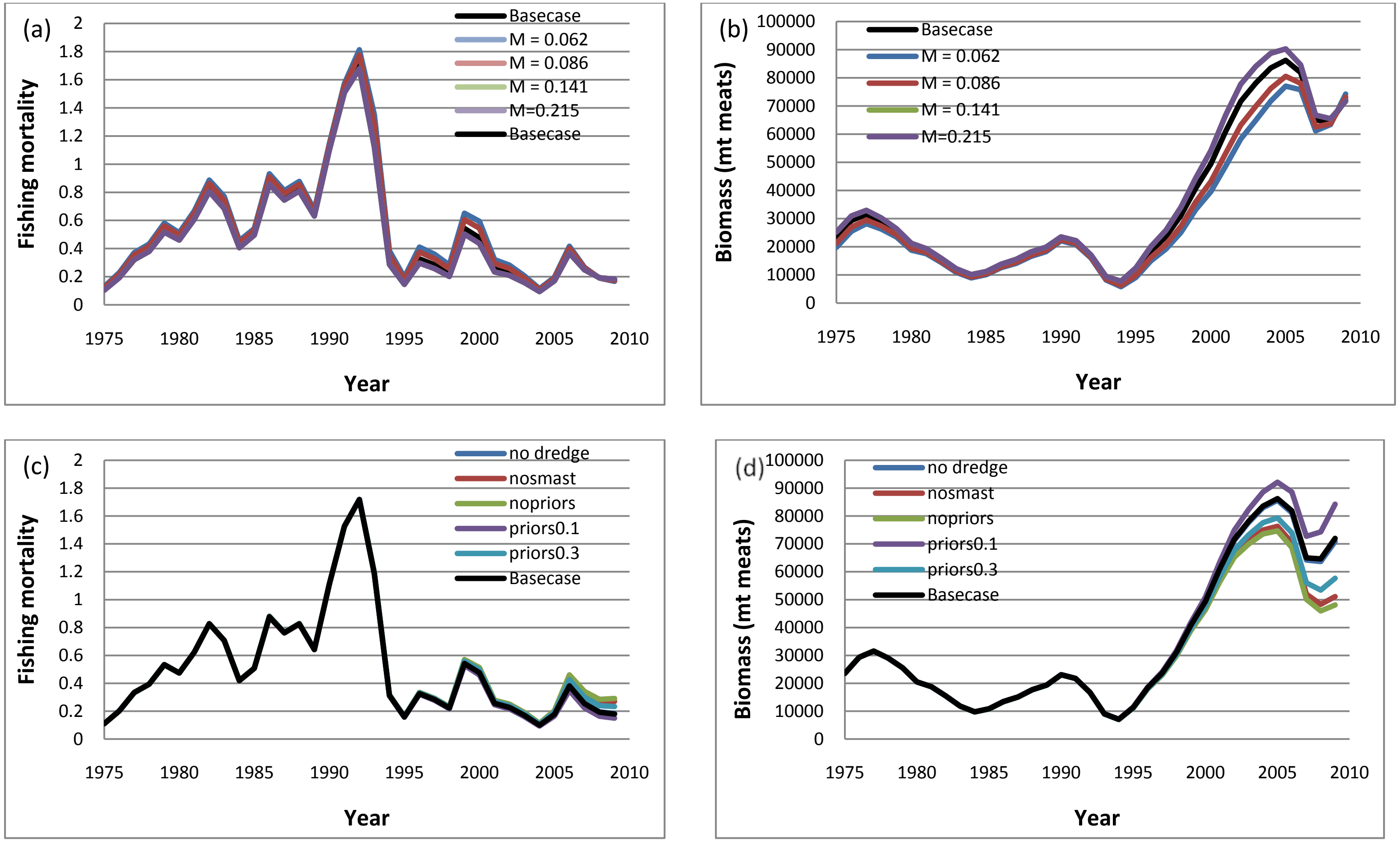


Figure B-51. Sensitivity analysis to the assumed value of (a-b) natural mortality and (c-d) priors on estimated values of fully recruited fishing mortality (a) and (c), and biomass (b) and (d) on Georges Bank. The values of natural mortality represent the assumed value (0.12, basecase) and the 5th, 25th, 75th and 95th percentile of the distribution of *M* used in the stochastic reference point model (Section 7). The assumptions on the priors are dredge and large camera cv = 0.15 (basecase), no dredge prior, no camera prior, no priors, cv = 0.1 for both priors and cv = 0.3 for both priors.

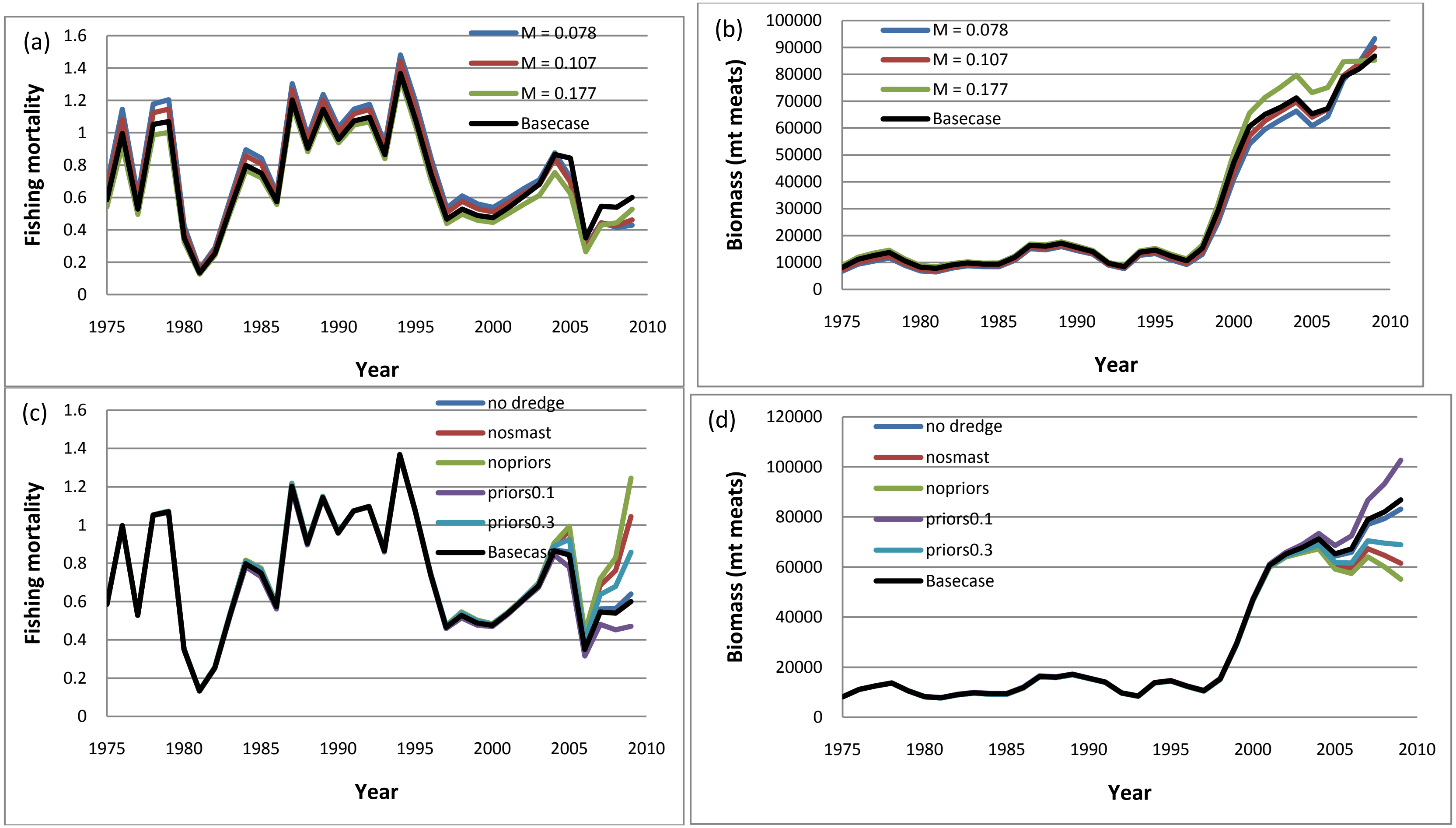
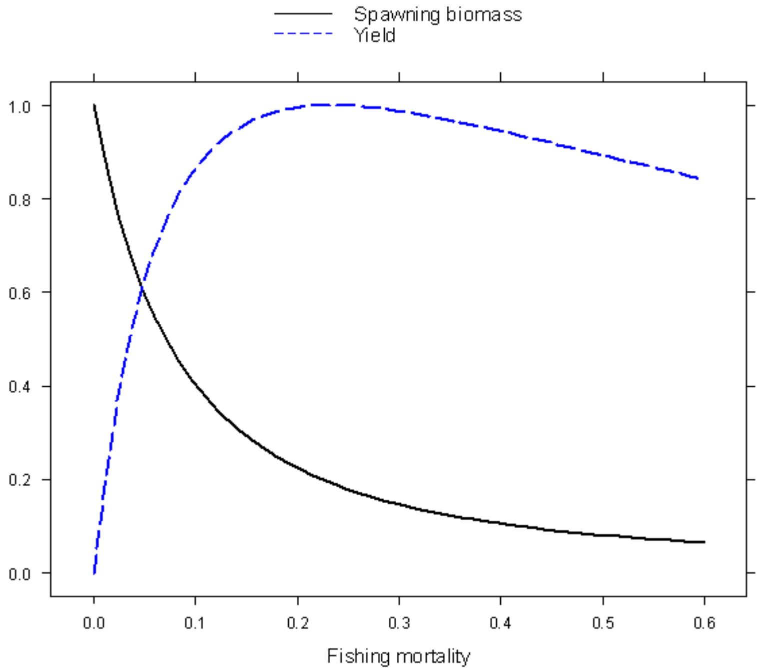


Figure B-52. Sensitivity analysis to the assumed value of (a-b) natural mortality and (c-d) large camera efficiency on estimated values of fully recruited fishing mortality (a) and (c), and biomass (b) and (d) on Georges Bank. The values of natural mortality represent the assumed value (0.12, basecase) and the 5th, 25th and 75th of the distribution of *M* used in the stochastic reference point model (Section 7; the model did not converge for the 95th percentile of M). The assumptions on the priors are dredge and large camera cv = 0.15 (basecase), no dredge prior, no camera prior, no priors, cv = 0.1 for both priors and cv = 0.3 for both priors.

(a)



(b)

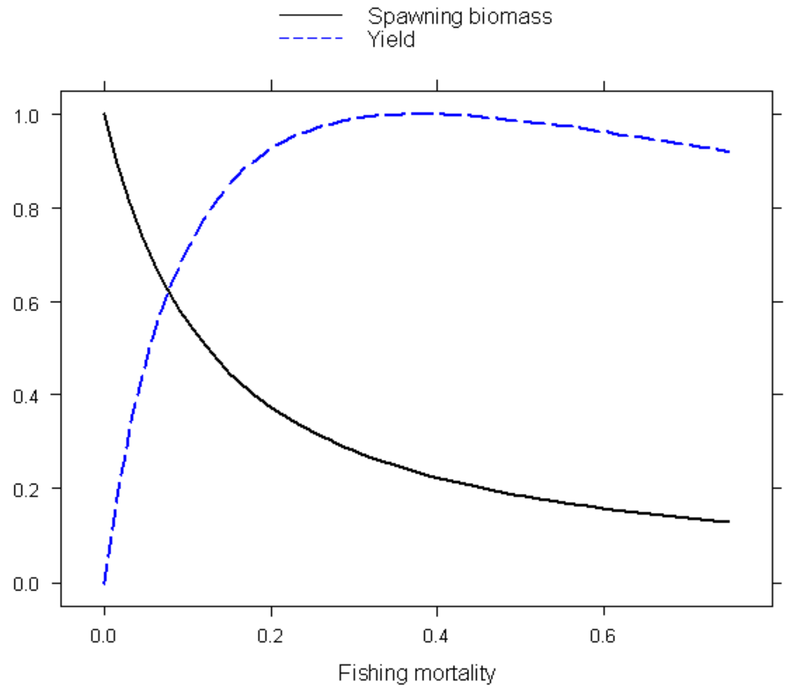
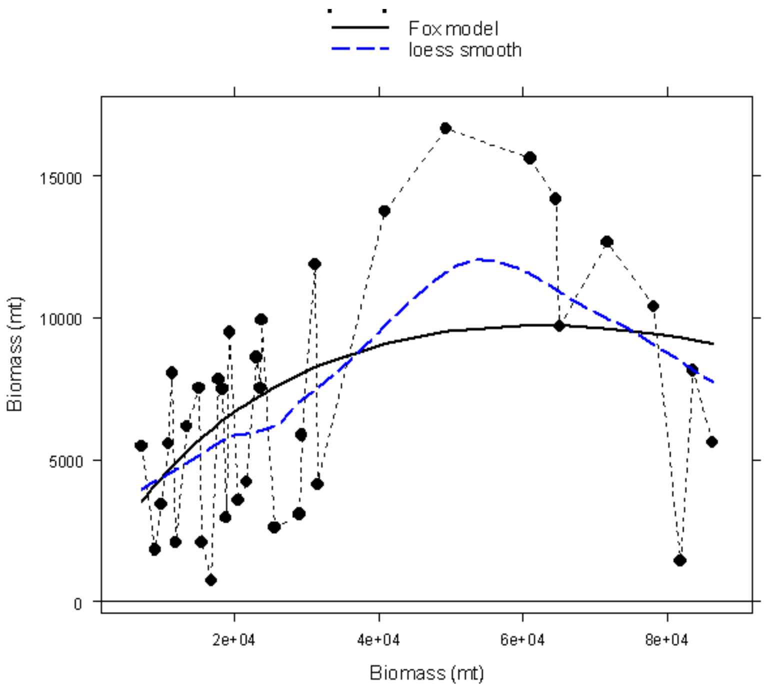


Figure B-53. Yield per recruit (blue dashed line) and spawning biomass per recruit (black solid line) for (a) Georges Bank and (b) Mid-Atlantic from the CASA model.

(a)



(b)

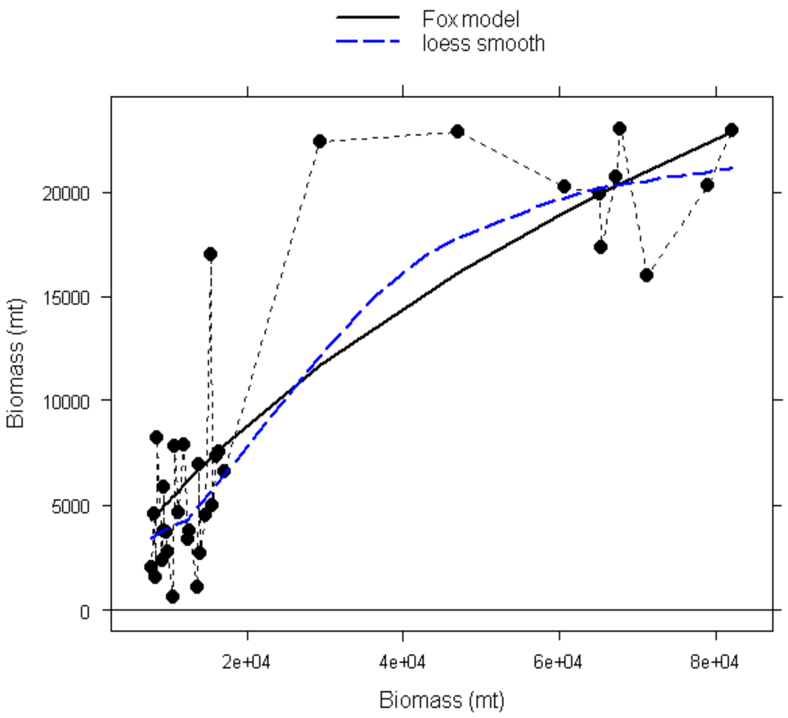


Figure B-54. Annual surplus production (solid circles) vs. biomass for (a) Georges Bank and (b) Mid-Atlantic. Fits to the Fox surplus production model (solid lines) and a lowess smoother are also shown.

(a) (b)

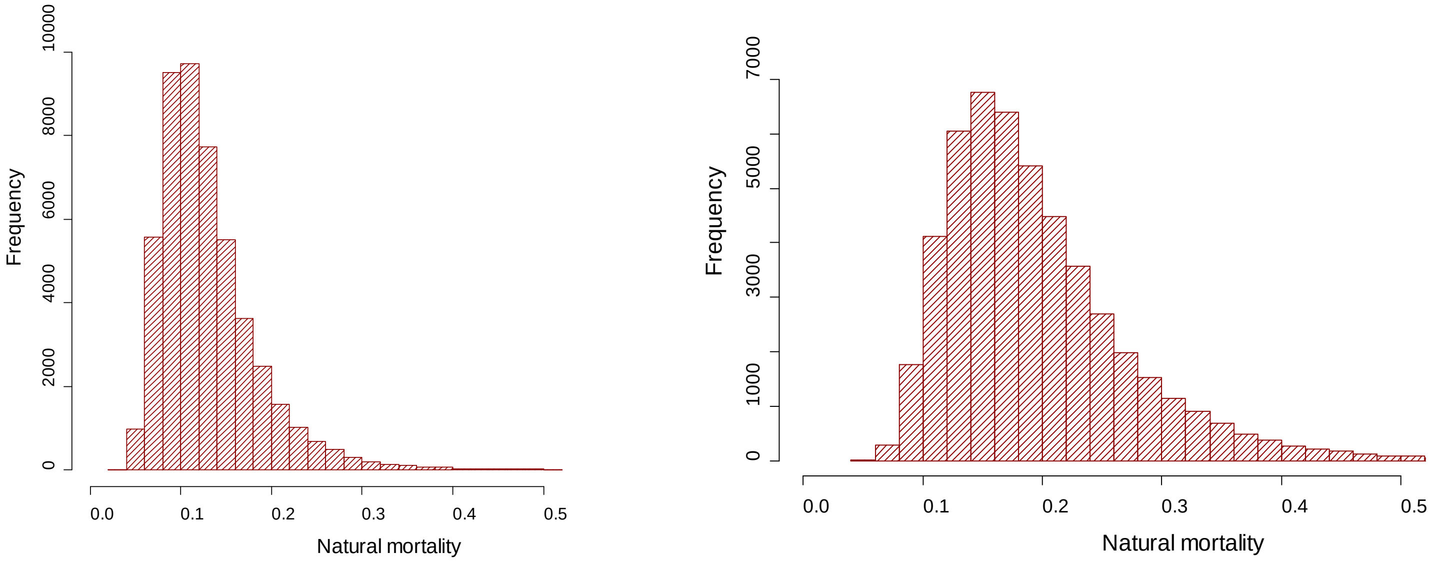
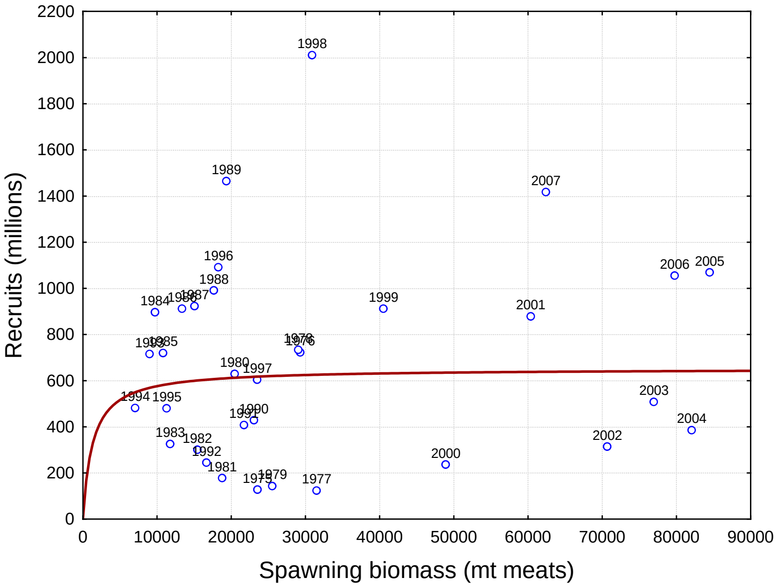


Figure B-55. Histograms of the assumed distributions of natural mortality in (a) Georges Bank and (b) the Mid-Atlantic.

(a)



(b)

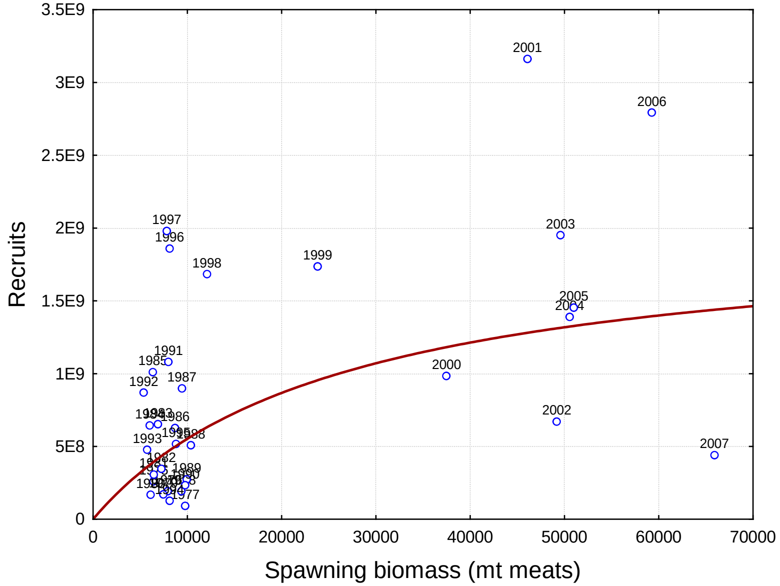


Figure B-56. Plots of stock-recruit relationships together with fits to Beverton-Holt stock-recruit curves for (a) Georges Bank and (b) Mid-Atlantic.

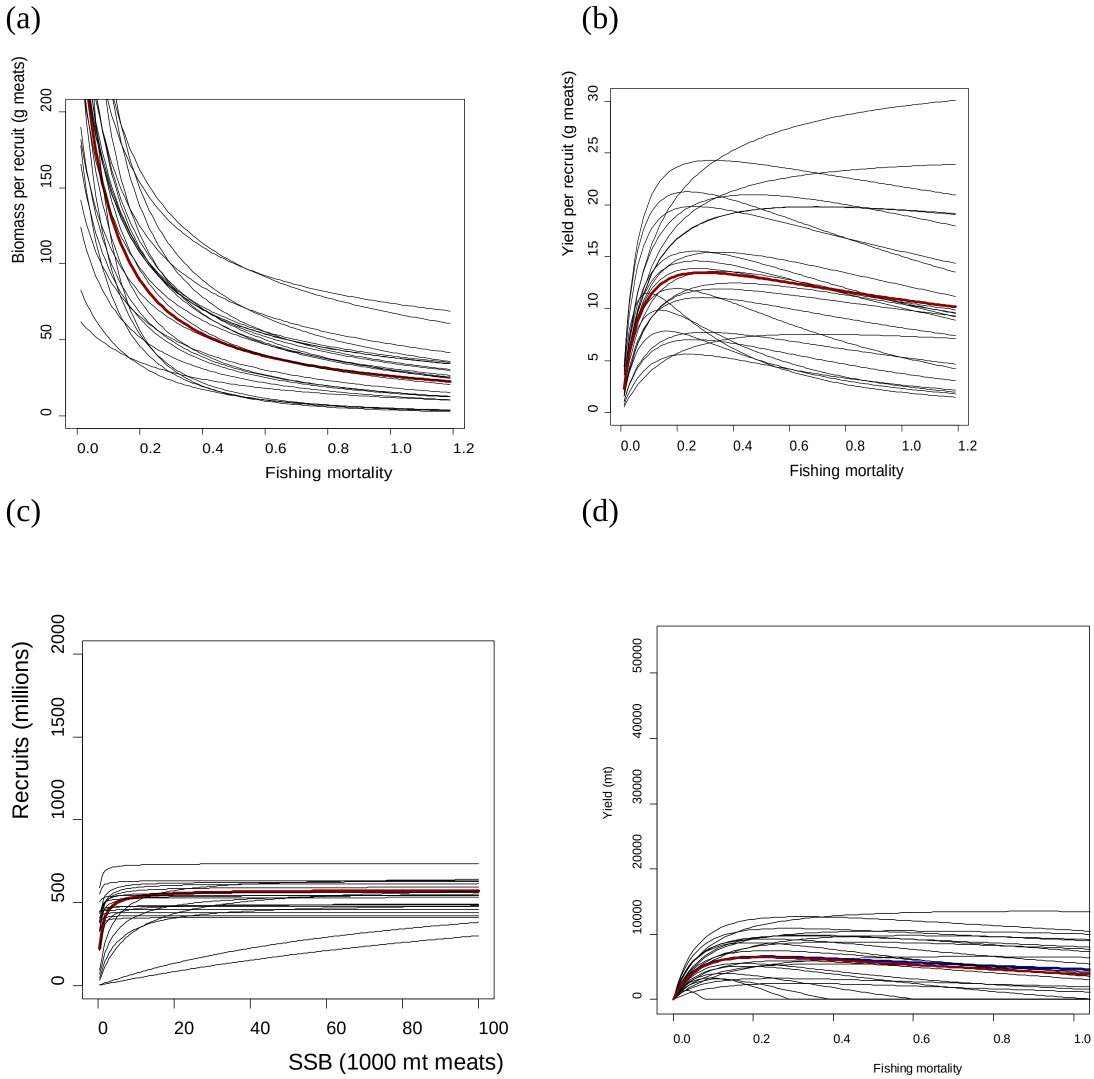


Figure B-57. Plots of (a) yield per recruit, (b) biomass per recruit, (c) stock-recruit and (d) yield from the SYM model for Georges Bank. The heavy red line is the mean of 50000 simulations, the blue line the median (yield only). 25 example plots from individual simulations are also shown.

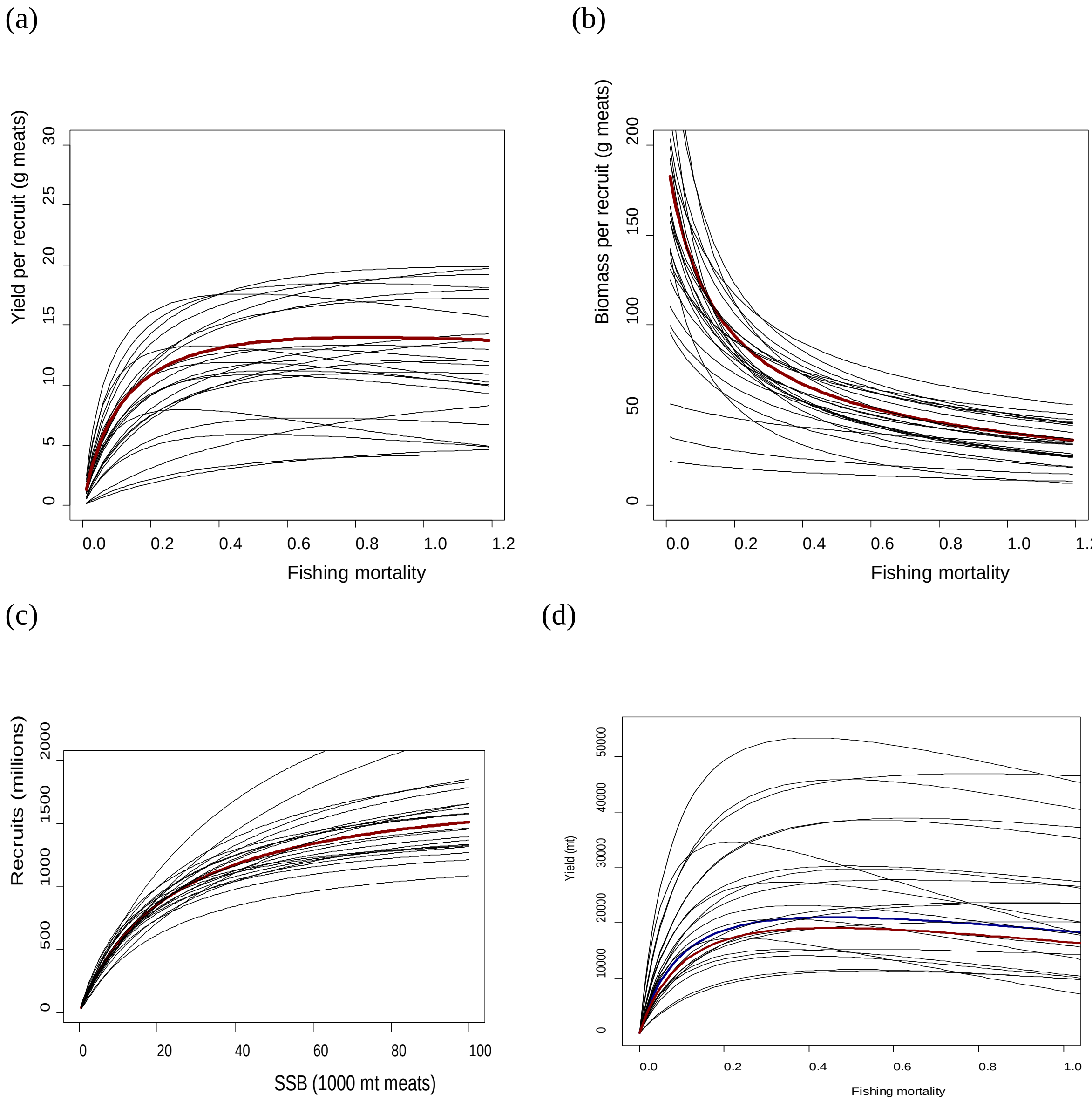


Figure B-58. Plots of (a) yield per recruit, (b) biomass per recruit, (c) stock-recruit and (d) yield from the SYM model for the Mid-Atlantic. The heavy red line is the mean of 50000 simulations, the blue line the median (yield only). 25 example plots from individual simulations are also shown.

(a)

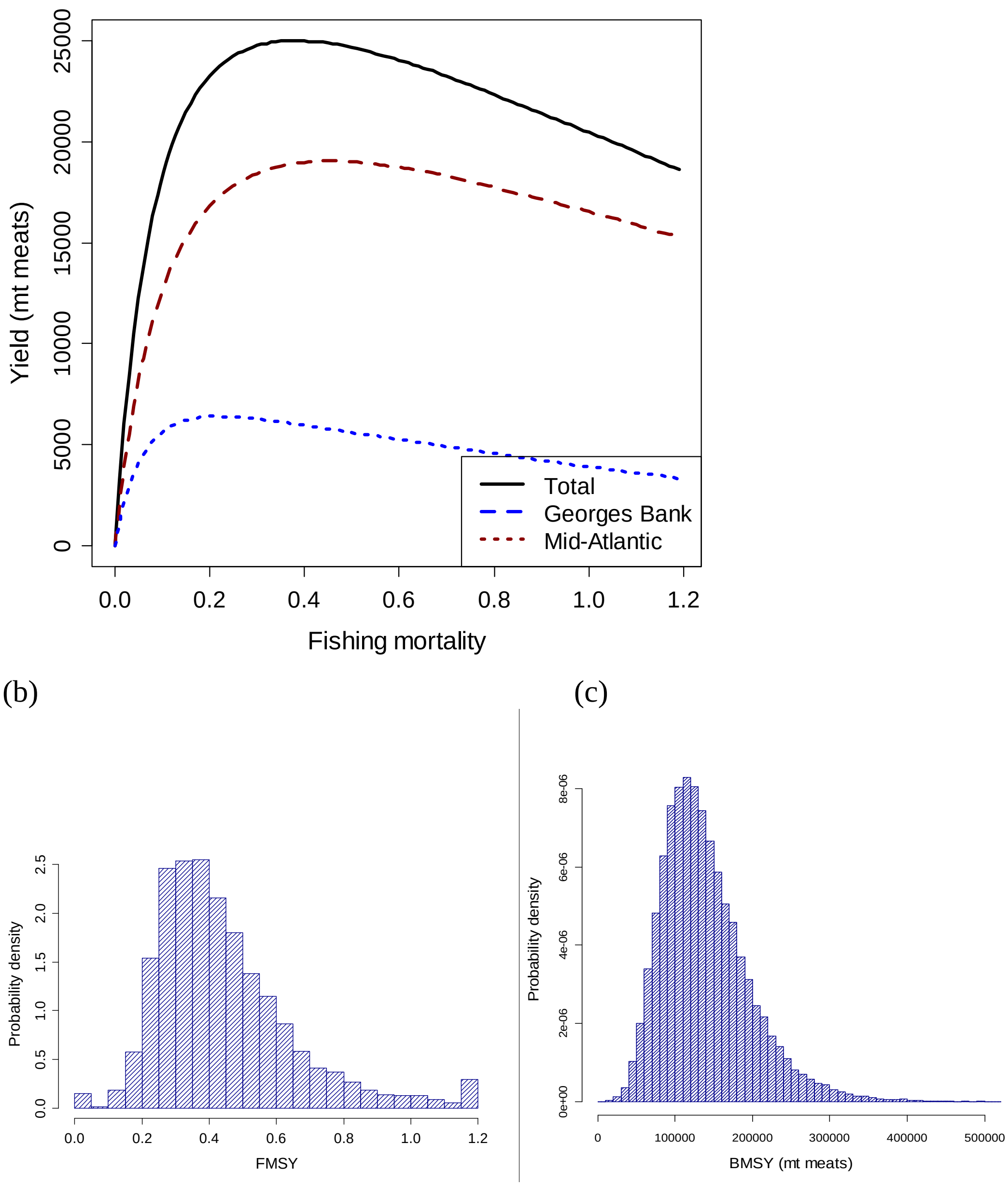


Figure B-59. (a) Median yield curves for Georges Bank, Mid-Atlantic, and overall yield. (b) Probability densities for whole-stock F_{MSY} and (c) probability densities for whole-stock B_{MSY} obtained from the SYM model.

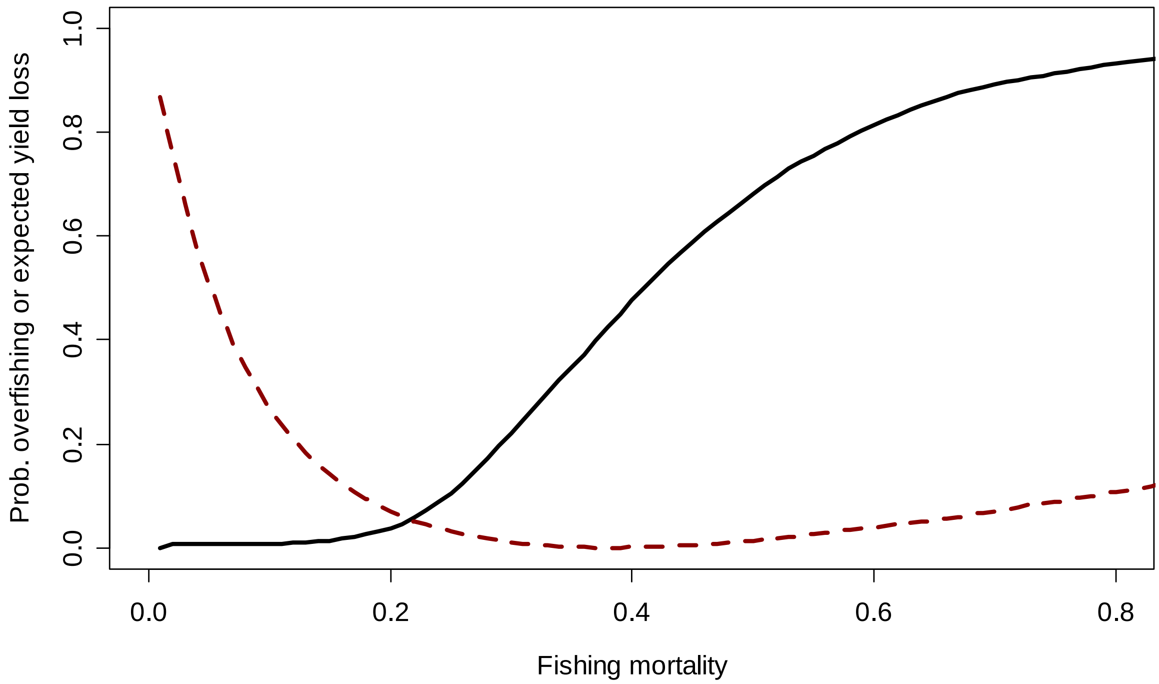


Figure B-60. The probability of overfishing as a function of realized fishing mortality (black solid line) and the loss of expected yield relative to that obtained at F_{MSY}.

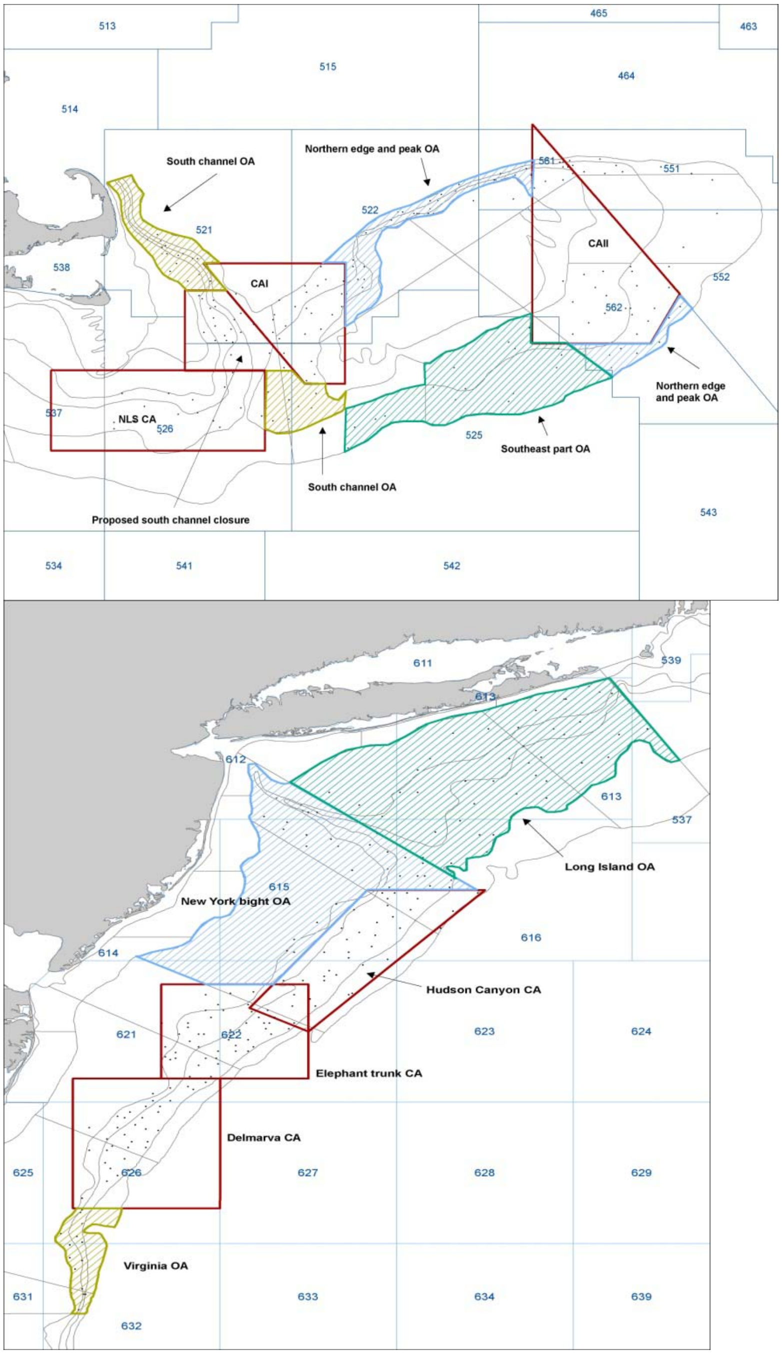


Figure B-61. Map of the 16 SAMS model areas. Each of the three Georges Bank closed areas are split into access and essential fish habitat areas, consistent with current management. Shellfish survey strata, NAFO statistical areas (rectangles), and 2009 NEFSC survey stations (dots) are also shown

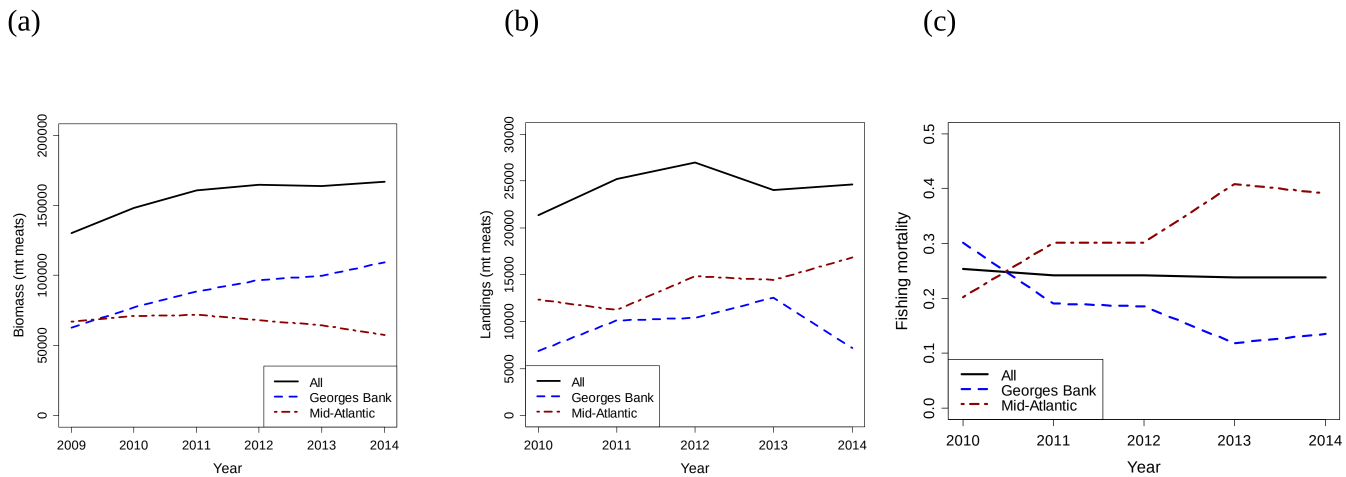


Figure B-62. Mean projected (a) biomass, (b) landings and (c) fishing mortality for Georges Bank (blue dashed line), Mid-Atlantic (red dot-dashed line) and overall (solid black line).

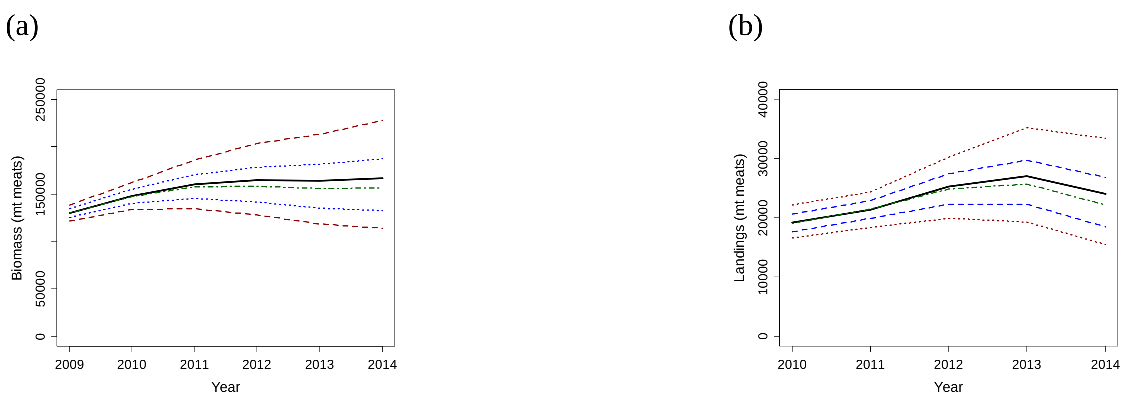


Figure B-63. The mean (black solid line), 10th and 90th percentiles (red dashed lines), 25th and 75th percentiles (dotted blue lines) and median (green dashed-dotted line) of projected overall (a) biomass and (b) landings.

Appendix B1: Invertebrate Subcommittee members

Invertebrate Subcommittee members who participated and contributed to the sea scallop assessment for SARC-50 at meetings during February-May, 2010. Participants are listed in alphabetical order by institution and then by last name.

Institution	Participants
Advanced Habitat Imaging Consortium (HabCam Group)	Karen Bolles, Patricia Keeting, Richard Taylor, Norman Vine
Department of Fisheries and Oceans, Halifax, N.S., Canada	Bob Mohn, Stephen Smith
Department of Fisheries and Oceans, Moncton, NB, Canada	Leslie-Anne Davidson
Fisheries Survival Fund	Ron Smolowitz
Maine Department of Marine Resources, Boothbay Harbour	Kevin Kelly
New England Fishery Management Council (NEFMC)	Andy Applegate, Deirdre Boelke, Jess Melgey
Northeast Fisheries Science Center, Woods Hole MA (NEFSC)	Jessica Blaylock, Larry Brady, Toni Chute, Dvora Hart (Assessment Lead), Daniel Hennen, Larry Jacobson (Invertebrate Subcommittee Chair), Chris Legault, Tim Miller, Victor Nordahl, Paul Rago, Alan Seaver, Jiashen Tang, Mark Terceiro
School for Marine Science and Technology, University of Massachusetts, Dartmouth (SMAST)	Jon Carey, Susan Inglis, Cate O'Keefe, Carly Mott, Kevin Stokesbury, Yuying Zhang
School of Marine Sciences, University of Maine, Orono	Sam Truesdell
Virginia Institute of Marine Science, College of William and Mary (VIMS)	Bill DuPaul, David Rudders
Woods Hole Oceanographic Institution (WHOI), Woods Hole, MA	Scott Gallager, Amber York

Appendix B2: Sea scallop discard estimates.

Jessica Blaylock, Northeast Fisheries Science Center, Woods Hole, MA.

Discard estimates for Atlantic sea scallop (*Placopecten magellanicus*) were calculated using the method described with the Standardized Bycatch Reporting Methodology (SBRM) (Wigley et al. 2007). This approach differs from that used in the previous assessment for this stock.

This paper presents updated sea scallop discard estimates for nine fleets, followed by a comparison of these values with the estimates presented at the previous assessment as part of the SAW 45 (NEFSC 2007).

Methods

Estimates of Atlantic sea scallop discards (mt meats) were derived for nine fleets: Georges Bank open and closed scallop dredge, and Mid-Atlantic Bight open and closed scallop dredge for the 1994 to 2009 time period, and Mid-Atlantic Bight scallop trawl, Georges Bank and Mid-Atlantic Bight small-mesh otter trawl, and Georges Bank and Mid-Atlantic Bight large-mesh otter trawl for the 1989 to 2008 time period.

In the scallop dredge analysis, observer and Vessel Trip Report (VTR) trips were partitioned into fleets using four classification variables: calendar quarter, gear type, area fished, and access area. In the scallop trawl and otter trawl analysis, observer and dealer trips were partitioned into fleets using the following four classification variables: calendar quarter, gear type, area fished, and mesh. Trips were not partitioned by trip category ('limited' versus 'general', for scallop dredge and scallop trawl) due to small sample size over the time series. Calendar quarter was based on landed date and used to capture seasonal variations in fishing activity. Gear type was based on Northeast gear codes (scallop dredge: negear 132; scallop trawl: negear 052; otter trawl: negear 050). Trips for which gear was unknown were excluded. Two broad geographical regions are defined for area fished based on statistical area: areas 520-562 constituted the Georges Bank (GBK) area, and areas 600 and above constituted the Mid-Atlantic Bight (MAB) area. For scallop dredge, two access area categories were used: ‘open’ and ‘closed’, where ‘closed’ includes all trips fishing in one of the scallop access areas (Closed Area I, Closed Area II and Nantucket Lightship in the GBK region; Hudson Canyon, Virginia Beach, Elephant Trunk, and Delmarva in the MAB region). Observer trips were assigned to the access area category based on program code, and VTR trips were assigned based on latitude and longitude. Finally, two mesh size groups were formed for otter trawl: small (mesh less than 5.5 inches) and large (5.5 inch mesh and greater).

Discards were estimated using a combined d/k_{all} ratio estimator (Cochran 1963), where d is discarded pounds of sea scallops and k_{all} is kept pounds of all species, calculated from Northeast Fishery Observer Program (NEFOP) data. Discard weight was derived by multiplying the d/k_{all} ratio of each fleet by the corresponding VTR or commercial landings (Wigley et al. 2007). Coefficients of variation (CV) were calculated as the ratio of the standard error of the discards divided by the discards.

In cases where limited observer data were available (i.e. two or less observed trips in a calendar quarter), an imputation approach was used to 'fill in' the missing (or incomplete) information using data from adjoining strata. In this imputation procedure, the temporal stratification (i.e., calendar quarter) was relaxed to entire year, recognizing that seasonal variations may occur that will thus not be accounted for. Numbers of annual observed trips by fleet are summarized in Table 1.

Comparison with previous discard estimates

Estimates of Atlantic sea scallop discards presented at SAW 45 were calculated for trips stratified by target species using the ratio of pounds of scallops caught for every pound of the target species landed (NEFSC 2007). Because of the different estimation method and stratification scheme, a direct comparison between these estimates and the current discard estimates was not possible.

To perform a more general comparison, sea scallop discard estimates from SAW 45 and current estimates were separated into two groups: estimated discards from trips using scallop gear (scallop dredge and scallop trawl), and estimated discards from trips using otter trawl gear. In the first group, scallop gear discard estimates from SAW 45 included the total 'estimated discards on directed scallop trips' (NEFSC 2007) for 1992-2006, which are assumed to be discards from trips that used mostly scallop dredge and scallop trawl gear. Scallop gear discard estimates from the current assessment included the sum of scallop dredge discard estimates across areas for the period 1994-2003, and the sum of scallop dredge and scallop trawl discard estimates across areas and gear types for the time period 2004-2006. For the second group, otter trawl discard estimates from SAW 45 included the total 'estimated discards in non-scallop otter trawl fisheries' (NEFSC 2007) for 1994-2006. Otter trawl discard estimates from the current assessment included the sum of otter trawl discard estimates across areas and mesh sizes for the period 1994-2006. For each group, a plot of SAW 45 sea scallop discard estimates and current estimates over time were produced. In addition, a third plot containing the sum of the SAW 45 estimates and the sum of current estimates described above was produced for illustrative purposes.

Discard to landings

To evaluate the proportion of estimated sea scallop discards to landings, the sum of the current discard estimates for scallop dredge was compared to the sum of estimated landings from Georges Bank, Southern New England, and Mid-Atlantic Bight (SAW 50) for the 1994 to 2009 time period.

Results and Discussion

Annual Atlantic sea scallop discard estimates by fleet are presented in Table 1 and Table 2. This analysis indicates that during the 1994 to 2008 time period, sea scallops were primarily discarded in the scallop dredge fleets with higher discarding in the 'open' category fleets. For 2008, estimated discards from the Mid-Atlantic Bight open and closed scallop dredge fleets were 201 and 52 mt meats, respectively. Estimated discards from the Georges Bank open and closed scallop dredge fleets were 214 and 96 mt meats, respectively. Discard estimates for the other five fleets for the same year ranged from less than 1 mt meats (Georges Bank small-mesh otter trawl) to 45 mt meats (Mid-Atlantic Bight large-mesh otter trawl).

The discard estimation presented here used a broad stratification approach. In addition, limitations are inherent in the use of VTR data for trip assignment to the 'access area' category because of missing or inaccurate position data. Consequently, these results should be considered as preliminary.

Comparison with previous discard estimates

Figure 1 shows updated sea scallop discard estimates compared with estimates presented at SAW 45. Accounting for missing estimates in some years (i.e. 2000, 2001), trends of discards are generally similar between the two sets of estimates. Values of discard estimates from trips using scallop gear are comparable between the two sets, while current estimate values from trips using otter trawl gear are lower than those presented at SAW 45 in most years. Both sets of sea scallop discard estimates indicate that a majority of discarding occurred in trips using scallop gear for the 1994 to 2006 time period.

Figure 2 indicates that trends in SAW 45 and current estimates are similar and resemble those observed with estimated sea scallop discards from trips using scallop gear (Figure 1A). This was expected, given the relatively small magnitude of the estimated discards from trips using otter trawl gear (Figure 1B).

These results provide an approximate comparison of current sea scallop discard estimates with those presented at SAW 45 and should be considered with caution given the different approaches used to obtain each set of estimates, as well as the missing estimates in some years. In particular, Figure 2 is meant to provide a general perspective of sea scallop discard estimates over the 1994 to 2006 time period, and exact values should not be used to convey total scallop discarding.

Discard to landings

Current estimates of discards and landings from 1994 to 2009 are presented in Figure 3. Total catch (discards plus landings) averaged 6,739 mt meats between 1994 and 1998. Catch increased in the following six years to peak at 31,348 mt meats in 2004, and averaged 26,490 mt meats from 2005 to 2009. Discards generally represent a small portion of total catch, with discard-to-landing ratios ranging from 0.006 in 1997 to 0.099 in 2003.

These results represent estimated sea scallop discards and landings in weight (mt meats). It is likely that discard-to-landing ratios of numbers would be higher because of the different size distribution of discarded scallops compared to that of landed scallops.

Acknowledgements

I wish to thank all the NEFOP observers for their diligent efforts to collect the discard information used in this analysis.

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Appendix B2-Table 1A. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) by the GBK open scallop dredge and GBK closed scallop dredge fleets, 1994-2009. Discards were not estimated for the GBK open scallop dredge fleet in 2000 and 2001 due to small sample size.

GBK open scallop dredge				GBK closed scallop dredge			
YEAR	Trips	Discards (mt meats)	CV	YEAR	Trips	Discards (mt meats)	CV
1994*	7	1	0.78	1994	n/a		
1995*	6	43	0.58	1995	n/a		
1996	15	103	0.37	1996	n/a		
1997*	11	26	0.67	1997	n/a		
1998*	9	6	0.46	1998	n/a		
1999*	8	51	0.68	1999*	15	53	0.26
2000	2			2000	226	246	0.03
2001	2			2001	16	28	0.16
2002*	11	100	0.39	2002	n/a		
2003*	14	177	0.45	2003	n/a		
2004*	16	34	0.32	2004	30	25	0.19
2005	41	372	0.36	2005	66	40	0.27
2006*	56	796	0.16	2006	79	41	0.26
2007	53	193	0.30	2007	127	41	0.26
2008	73	201	0.23	2008	140	52	0.12
2009	58	265	0.33	2009	23	24	0.30

** Imputed data were used for discard estimation for these years.*
n/a: not applicable

Appendix B2-Table 1B. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) by the MAB open scallop dredge and MAB closed scallop dredge, 1994-2009. Discards were not estimated for the MAB open scallop dredge fleet in 2001 due to small sample size.

MAB open scallop dredge				MAB closed scallop dredge			
		Discards				Discards	
YEAR	Trips	(mt meats)	CV	YEAR	Trips	(mt meats)	CV
1994	16	276	0.59	1994	n/a		
1995*	20	341	0.28	1995	n/a		
1996	23	22	0.72	1996	n/a		
1997*	18	8	1.15	1997	n/a		
1998*	16	42	0.66	1998	n/a		
1999*	8	7	0.56	1999	n/a		
2000	28	749	0.33	2000	n/a		
2001	3			2001	85	301	0.09
2002*	13	1,446	0.19	2002	74	150	0.10
2003	62	2,206	0.14	2003	46	119	0.12
2004	143	1,856	0.13	2004	92	503	0.10
2005	166	367	0.29	2005	54	38	0.21
2006*	87	71	0.39	2006*	6	3	0.49
2007	85	66	0.40	2007	93	63	0.22
2008	89	214	0.54	2008	337	96	0.14
2009	118	549	0.16	2009	233	199	0.16

** Imputed data were used for discard estimation for these years.*
n/a: not applicable

Appendix B2-Table 1C. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) by the MAB scallop trawl fleet, 1989-2008. Discards were not estimated prior to 2004 due to small sample size.

MAB scallop trawl			
YEAR	Trips	Discards (mt meats)	CV
1989			
1990			
1991			
1992			
1993			
1994			
1995			
1996			
1997			
1998			
1999			
2000			
2001	4		
2002	1		
2003			
2004*	44	99	0.25
2005	137	61	0.13
2006*	30	150	0.33
2007	34	15	0.58
2008*	38	7	0.61

** Imputed data were used for discard estimation for these years.*

Appendix B2-Table 1D. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) by the GBK small-mesh otter trawl, and MAB small-mesh otter trawl fleets, 1989-2008.

GBK small-mesh otter trawl				MAB small-mesh otter trawl			
YEAR	Trips	Discards (mt meats)	CV	YEAR	Trips	Discards (mt meats)	CV
1989	64	2	0.53	1989	34	213	0.39
1990	31	<1	1.22	1990	47	8	0.44
1991	68	<1	0.80	1991	78	11	2.05
1992	42	<1	0.69	1992	47	6	0.53
1993*	15	<1	0.79	1993*	7	14	1.12
1994*	10	23	0.80	1994*	11	41	1.03
1995*	10	<1	1.40	1995	63	71	0.23
1996*	11	0	0.00	1996	78	15	1.62
1997*	19	<1	0.88	1997*	49	1	2.83
1998*	5	<1	1.61	1998*	26	5	1.43
1999*	8	<1	2.62	1999	33	21	1.07
2000*	17	<1	0.49	2000	34	2	0.95
2001*	14	<1	0.64	2001	54	<1	8.88
2002*	33	<1	0.81	2002	32	68	0.34
2003	54	<1	1.10	2003	72	18	0.75
2004	107	2	0.99	2004	246	6	0.38
2005	191	<1	0.47	2005	166	4	0.33
2006	59	<1	0.55	2006	144	14	2.50
2007	62	<1	1.54	2007	216	5	0.55
2008	49	<1	0.48	2008	149	10	0.54

** Imputed data were used for discard estimation for these years.*

Appendix B2-Table 1E. Number of observed trips, sea scallop discards (mt meats) and coefficient of variation (CV) by the GBK large-mesh otter trawl, and MAB large-mesh otter trawl fleets, 1989-2008. Discards were not estimated for MAB large-mesh otter trawl prior to 1992 due to small sample size.

GBK large-mesh otter trawl				MAB large-mesh otter trawl			
YEAR	Trips	Discards (mt meats)	CV	YEAR	Trips	Discards (mt meats)	CV
1989	27	1	0.88	1989	1		
1990	33	1	0.72	1990			
1991	34	4	0.54	1991	4		
1992	35	<1	1.10	1992*	14	4	0.40
1993*	22	3	0.60	1993*	7	0	0.00
1994	27	<1	1.24	1994*	13	230	0.57
1995	60	<1	0.42	1995	52	107	0.80
1996	33	<1	0.78	1996*	16	<1	0.57
1997*	21	<1	1.00	1997*	5	0	0.00
1998*	7	<1	0.67	1998*	13	3	1.79
1999*	11	<1	1.36	1999*	5	0	0.00
2000	26	<1	0.54	2000	27	9	1.54
2001	51	<1	0.45	2001*	44	10	1.02
2002	77	2	0.60	2002*	37	8	2.37
2003	161	3	0.76	2003*	11	42	0.92
2004	314	42	0.35	2004	91	19	0.32
2005	952	10	0.18	2005	87	2	0.80
2006	457	30	0.37	2006	63	16	0.72
2007	463	5	0.25	2007	160	13	0.54
2008	562	6	0.21	2008	127	45	1.02

** Imputed data were used for discard estimation for these years.*

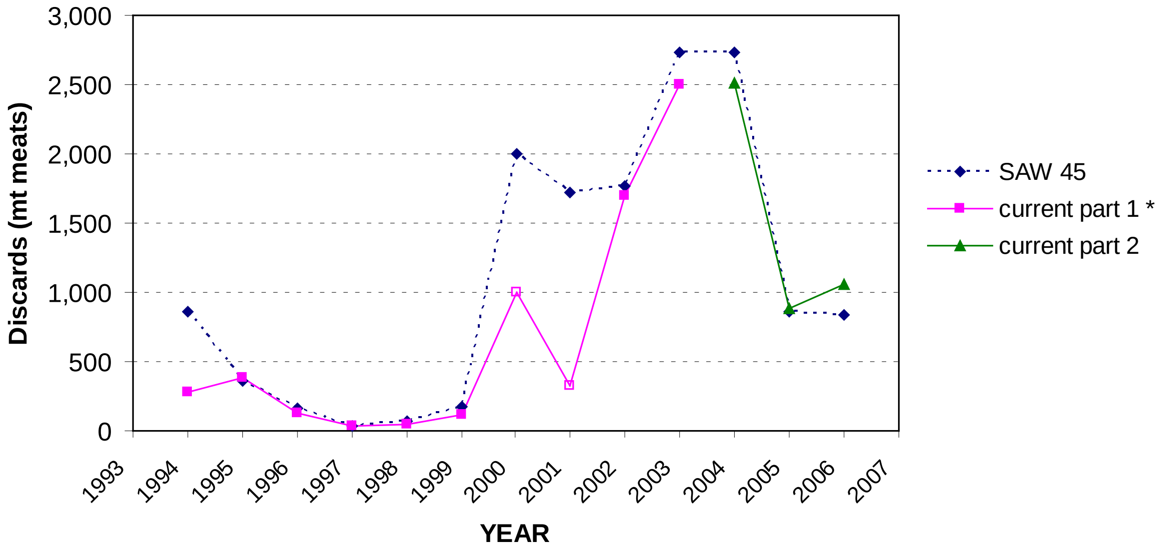
Appendix B2-Table 2. Summary of sea scallop discard estimates (mt meats) from Table 1 by region, 1994-2009.

Georges Bank (GBK)						Mid-Atlantic Bight (MAB)						
YEAR	open scallop dredge	closed scallop dredge	small-mesh otter trawl	large-mesh otter trawl	Total	YEAR	open scallop dredge	closed scallop dredge	scallop trawl	small-mesh otter trawl	large-mesh otter trawl	Total
1994	1	n/a	23	1	24	1994	276	n/a	*	41	230	547
1995	43	n/a	0	0	43	1995	341	n/a	*	71	107	519
1996	103	n/a	0	0	103	1996	22	n/a	*	15	1	38
1997	26	n/a	0	0	26	1997	8	n/a	*	1	0	9
1998	6	n/a	0	0	6	1998	42	n/a	*	5	3	50
1999	51	53	0	0	104	1999	7	n/a	*	21	0	28
2000	*	246	0	1	247	2000	749	n/a	*	2	9	760
2001	*	28	1	0	29	2001	*	301	*	1	10	312
2002	100	n/a	0	2	103	2002	1,446	150	*	68	8	1,673
2003	177	n/a	0	3	181	2003	2,206	119	*	18	42	2,386
2004	34	25	2	42	103	2004	1,856	503	99	6	19	2,482
2005	372	40	0	10	421	2005	367	38	61	4	2	473
2006	796	41	1	30	868	2006	71	3	150	14	16	254
2007	193	41	0	5	240	2007	66	63	15	5	13	162
2008	201	52	0	6	259	2008	214	96	7	10	45	372
2009	265	24	+	+	289	2009	549	199	+	+	+	748

** No discard estimate due to small sample size.*

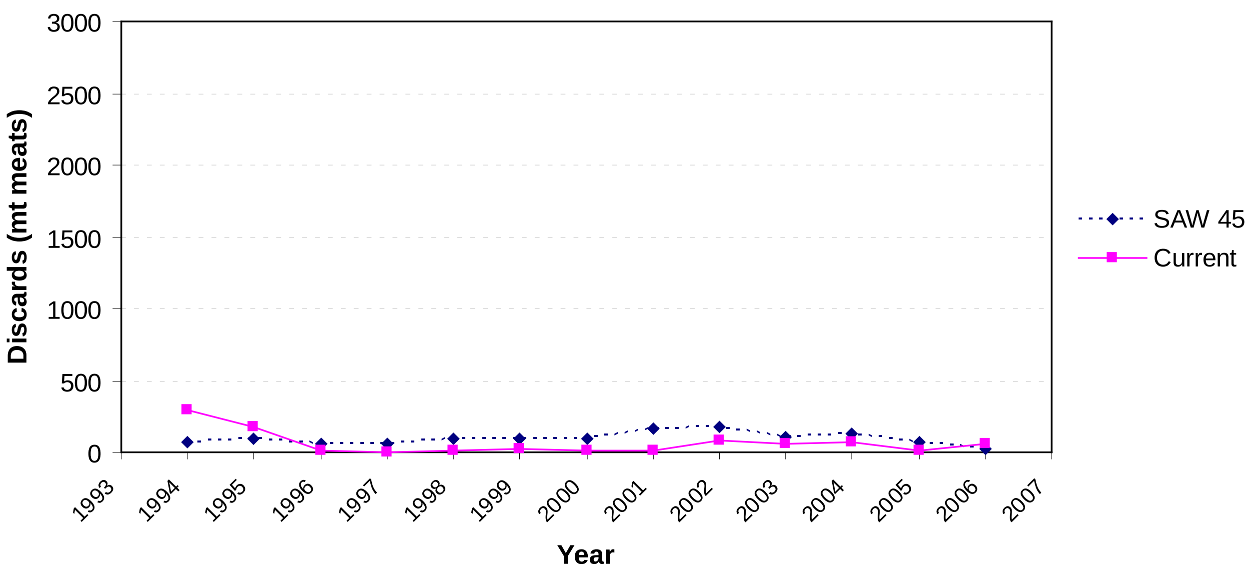
+ No discard estimate because 2009 data not yet available.

n/a: not applicable

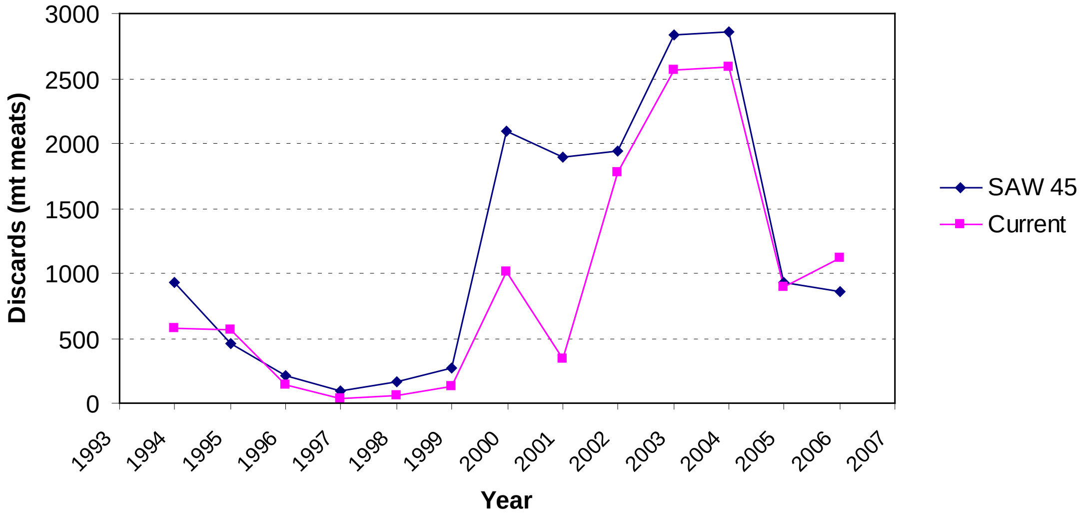


** Discards in 2000 and 2001 are underestimates because no discards were estimated for GBK open scallop dredge in 2000 and 2001, and MAB open scallop dredge in 20001 due to small sample size.*

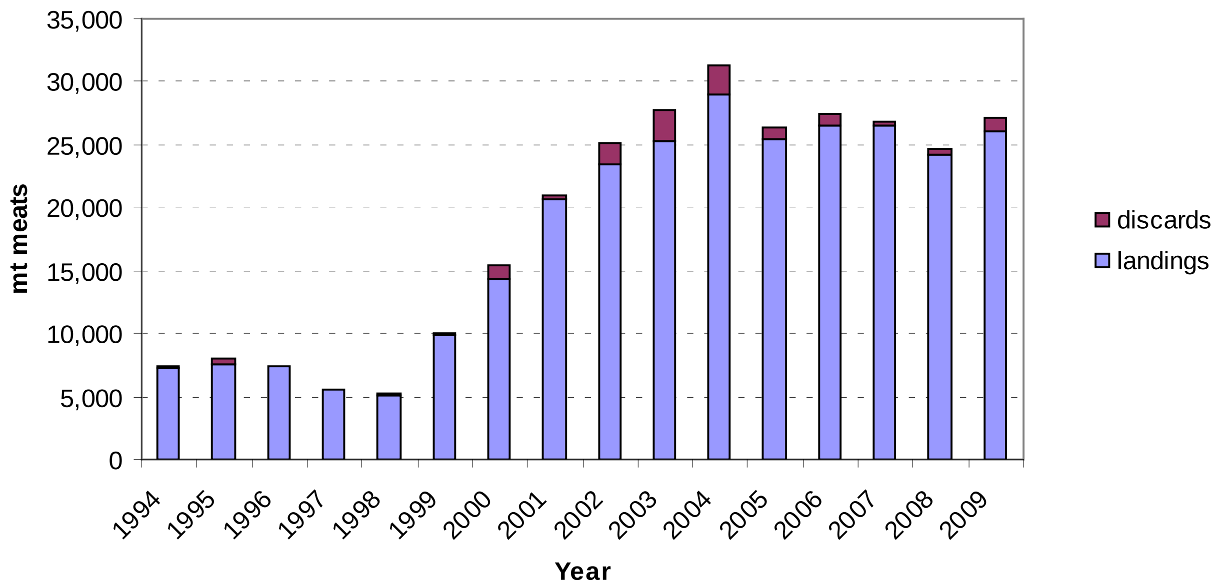
Appendix B2-Figure 1A. SAW 45 and current estimated sea scallop discards from trips using scallop gear (mt meats), 1994-2006. Current part 1 includes estimates from scallop dredge trips only, while part 2 includes estimates from scallop dredge and scallop trawl.



Appendix B2-Figure 1B. Estimated sea scallop discards from trips using otter trawl gear (mt meats), 1994-2006.



Appendix B2-Figure 2. Sum of the SAW 45 and current estimated sea scallop discards (mt meats) presented in Figure 1 (estimated discards from trips using scallop gear, and estimated discards from trips using otter trawl gear), 1994-2006. This figure is for illustration purposes only and is not meant to convey exact total sea scallop discard estimates.



Appendix B2-Figure 3. Estimated scallop landings (SAW 50) and current estimated sea scallop discards from scallop dredge fleets (mt meats), 1994-2009.

Appendix B3: Comparison of scallop density estimates using the SMAST scallop video survey data with a reduced view field and reduced counts of individuals per image.

O'Keefe, Catherine E.[1], Jon D. Carey[1], Larry D. Jacobson[2], Deborah R. Hart[2] and Kevin D.E. Stokesbury[1]

[1]University of Massachusetts Dartmouth
School for Marine Science and Technology
200 Mill Road, Suite 325
Fairhaven, MA 02719

[2]National Marine Fisheries Service
Northeast Fisheries Science Center
166 Water Street
Woods Hole, MA 02543

Introduction

The University of Massachusetts Dartmouth School for Marine Science and Technology (SMAST) has conducted an annual continental shelf-wide video survey for scallops in the Mid-Atlantic and Georges Bank areas since 2003. The survey provides information about abundance, density, shell height distribution and spatial aggregation of scallops in the Mid Atlantic Bight (MAB) and Georges Bank (GBK) regions of the scallop resource.

In this analysis, we examined alternative methods for calculating scallop density from SMAST survey images. To address potential bias in density calculations resulting from scallops on an edge of the visible image, we compared different methods of counting scallops and different methods for expanding the image view area. For this assessment, the Invertebrate Subcommittee decided to calculate density using all scallops visible in images (as before) and an assumed view field equal to the area calculated from the dimensions of the sample frame plus ½ of mean shell height in each area for each year. This increased density estimates by 1-3% in the MAB and GBK stock areas. In the future, densities will be calculated both by excluding scallops that lie on the top and right edges of the video images and using the area within the sampling frame and by including all visible scallops and adjusting the dimensions of the sample frame based on mean shell by area.

Methods

Original densities for the Mid Atlantic Bight and Georges Bank scallop stocks were calculated according to Stokesbury (2002) and Stokesbury et al. (2004). All scallops in each image were counted. The large camera view area of $2.84m^2$ (1.986m x 1.430m) was expanded to account for scallops that were positioned on the edges of the image. The expansion of the view area was calculated based on a mean shell height of 112mm as observed in the 1999 Nantucket Lightship Closed Area video survey. We added half of the mean shell height to each edge of the camera view field to expand the area to $3.235m^2$ ((1.986m + (2*56mm))*(1.430 + (2*56mm)), see Figure 1. Mean densities and standard errors are calculated according to Cochran (1977) for a two-stage sampling design. Density estimates represent the mean of the mean scallops per station, where there are 4 quadrats per station. The mean of the total sample is:

(1) $$\overline{\overline{x}} = \sum_{i=1}^{n} \left(\frac{\overline{x}_i}{n} \right)$$

where:

n = primary sample units (stations)

$\overline{x}_i$ = sample mean per element (quadrat) in primary unit i (stations)

$\overline{\overline{x}}$ = the grand mean over the two stages of sampling.

The standard error of this mean is approximately:

(2) $$S.E.(\overline{x}) = \sqrt{\frac{1}{n}(s^2)}$$

where:

$s^2 = \sum^{n} (\overline{x}_i - \overline{\overline{x}})^2 / (n-1)$ = variance among primary unit (stations) means.

This simplified version of the two-stage variance is possible when the sampling fraction n/N is small, hundreds of m^2 are sampled compared with millions of m^2 in the area (Cochran, 1977; Krebs, 1999).

Experimental evaluation

We examined density estimates in a sample of images based on removing scallop counts from two edges of the image and not including an expansion adjustment to the image. By removing the counts from two edges of the image, the scallop counts are independent of scallop shell height. Counting scallops on only one vertical and one horizontal edge of each image reduces potential bias for inclusion of a greater number of small scallops than large scallops. We analyzed images from the Elephant Trunk Closed Area (ETCA) between 2003 and 2009 from our broad-scale 3 nm video survey. We counted scallops in the image with any portion of the animal along the top and right edges of the image. We subtracted the counts of the animals on these edges from the total count of animals in the image. We calculated density based on the actual camera view field without any expansion factor ($2.84m^2$; Table 1). This method of calculation is consistent with land-based ecological methods (Krebs, 1999). Results showed that densities calculated in this manner were slightly higher than the original estimates. Interestingly, the decreases in numbers counted tends to be offset by the increases in area resulting in a slight increase in calculated density.

Ratio estimator approach for potential use in this assessment

We also used a ratio estimator (Cochran, 1977; Krebs, 1999) to determine the relative difference in densities between the original and reduced count density calculations. Again, we examined 2003-2009 ETCA scallop data (Table 1). The ratio estimator for the original densities and the densities that excluded scallop counts on two edges of the image is:

(3) $$\hat{R} = \frac{y}{x}$$

where:

y = reduced mean density
x = original mean density.

Historical density data might be adjusted approximately using the ratio estimate (i.e. adjusted density = *dR*). We calculated the variance in adjusted density estimates using an exact formula for the ratio of two independent variables (Goodman, 1960):

(4) $$V(dR) = d^2V(R) + R^2V(d) + V(d)V(R)$$

where:

$V(x)$ = variance of x
d = original density estimate
R = ratio estimate

We also pooled the data for all years and calculated the ratio between the original scallop counts and the counts that excluded the top and right edges. We then applied this overall ratio to each year to calculate a new density for each year (Table 1). We calculated variance in the same way as the individual year variance estimates.

Expanded area approach for potential use in this assessment

Finally, we examined an alternative approach to density calculations that incorporated an image view expansion based on shell height by area. We determined the annual mean shell height for 2003 - 2009 in ETCA and recalculated density estimates by changing the camera view area adjustment. Instead of using the Nantucket Lightship Closed Area 1999 mean shell height (112mm) as a constant for expanding the camera view field, we used the mean shell height by area, by year (Figure 2). The camera view field expansion varied by year based on the equation:

(5) $$ViewArea = (1.986m + (2*(\frac{meanSH}{2}))*(1.430m + (2*(\frac{meanSH}{2}))$$

where:

$meanSH$ = mean shell height.

For this analysis, we included all scallop counts and calculated the mean of the mean scallops per station, where each station had 4 quadrats (Table 1). The adjusted view field method did not include increases in variance so that uncertainty in the adjusted figures may be understated.

Comparison of methods applied to the same sample data

We applied the ratio and camera view field adjustments to the video survey data for the Mid Atlantic Bight (MAB) and Georges Bank (GBK) 3 nm survey estimates from 2003 through 2009. We compared the original density estimates with the overall ratio adjusted estimate and the mean shell height adjusted camera view area adjustment (Table 2).

Results

Table 1 and Figure 3 show a comparison of the original, yearly ratio adjusted, overall ratio adjusted and shell height adjusted density estimates for the ETCA from 2003-2009.

Tables 2 and 3 show the mean shell height adjusted density, abundance and biomass estimates for the MAB and GBK scallop resource areas from 2003-2009 for large and small cameras. On average, the density estimates increased by 1-3%.

Conclusion

It would have been ideal to reexamine all video images collected during 2003-2009 to exclude sea scallops along two edges of the view field from counts, and compute densities using the actual area of the sample frame, but this was not possible in time for the assessment. The only practical alternatives were to use either the ratio estimators or adjusted view field approach to correct the overall densities for each region and year.

The Invertebrate Subcommittee considered both approaches and decided to use the adjusted view field method because it accommodated differences among years in mean shell height, which may be important. For adjusting the stock assessment data, the adjusted view field approach was based on the average size of sea scallops in each area and year for the survey as a whole, rather than the average size in each image. The two types of adjustment factors were similar but no rigorous comparison of the two approaches was carried out.

Future research will include counting scallops that lie on the top and right edges of the image and subtracting those counts from the count of total scallops in the image. We will compare density estimates that include all counts with the reduced count estimates.

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Appendix B3-Table 1. ETCA 2003-2009 Large camera original density, reduced count density, ratio adjusted density (reduced count/original count), overall ratio adjusted density and mean shell height adjusted density.

Year	Mean SH	SH Adj Area	Original Density	Reduced Count Density	Ratio Adj Density (R/O)	Overall Ratio Adj Density	SH Adj Density
2003	60	3.049	2.1859	2.4463	2.4463	2.3848	2.3196
2004	81	3.123	0.8507	0.9426	0.9426	0.9281	0.8812
2005	94	3.170	0.7485	0.8156	0.8156	0.8166	0.7638
2006	98	3.184	0.6336	0.6656	0.6656	0.6913	0.6437
2007	98	3.183	0.5965	0.6438	0.6438	0.6508	0.6063
2008	101	3.194	0.4934	0.5288	0.5288	0.5383	0.4998
2009	99	3.187	0.1813	0.1986	0.1986	0.1978	0.1840

Appendix B3-Table 2. Large camera area surveyed, mean shell height (mm), shell height adjusted view field (m^2), shell height adjusted density, abundance, biomass and 95% CI of the density for the MAB and GBK stock areas from 2003-2009.

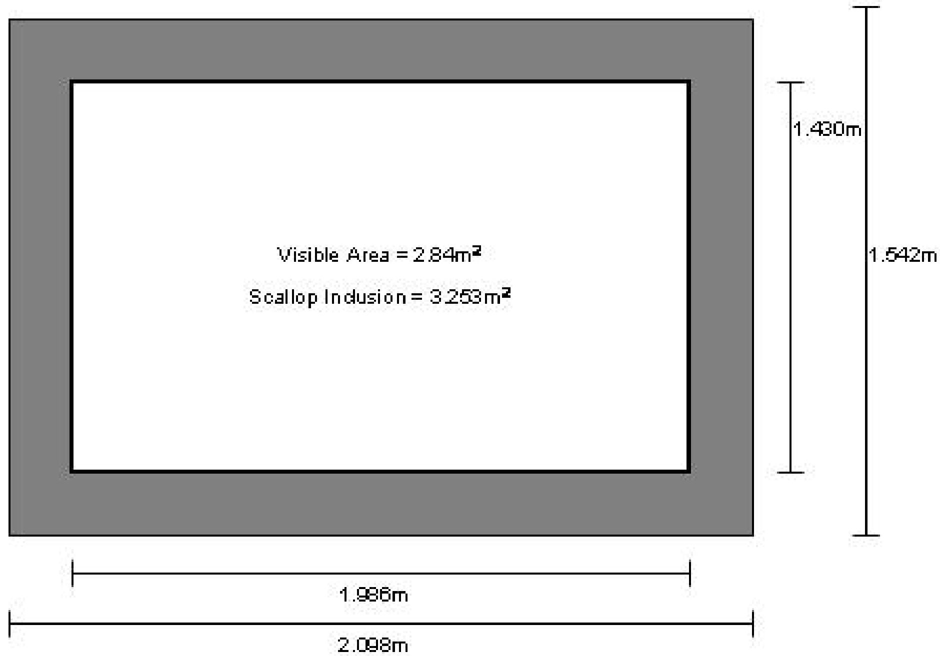
LARGE CAMERA

Year	Stations	Area Surveyed km2	Mean SH mm	SH Adj Area m2	SH Adj Density sc/m2	Abundance	Biomass (mt)	95% CI
				Mid Atlantic Bight				
2003	804	**24664**	73.9	3.098	0.5047	12525017415.1	113401.8	0.16
2004	840	**25591**	90.4	3.157	0.2293	5945022074.1	80569.1	0.04
2005	864	**26547**	91.6	3.161	0.2148	5729979610.2	86770.8	0.05
2006	897	**26918**	92.0	3.163	0.1954	5411614262.4	78088.9	0.04
2007	941	**28739**	94.5	3.172	0.1826	5305430005.0	80333.9	0.03
2008	931	**28184**	91.4	3.161	0.1883	5412596845.3	85561.1	0.04
2009	928	**28647**	96.4	3.179	0.1366	3913262600.8	64727.5	0.02
				Georges Bank				
2003	929	**27906**	102.1	3.199	0.1486	4260307453.6	89080.9	0.02
2004	935	**28430**	107.5	3.219	0.1223	3528997219.0	82852.3	0.03
2005	902	**27844**	106.6	3.215	0.1169	3254941556.5	76277.7	0.02
2006	939	**28276**	114.6	3.245	0.1093	3167772661.9	89942.1	0.02
2007	912	**27813**	99.0	3.188	0.1438	4047458860.7	87482.7	0.03
2008	910	**27227**	93.3	3.167	0.0998	2804734412.4	48591.2	0.02
2009	899	**29079**	92.2	3.164	0.1603	4448902027.8	72959.5	0.03

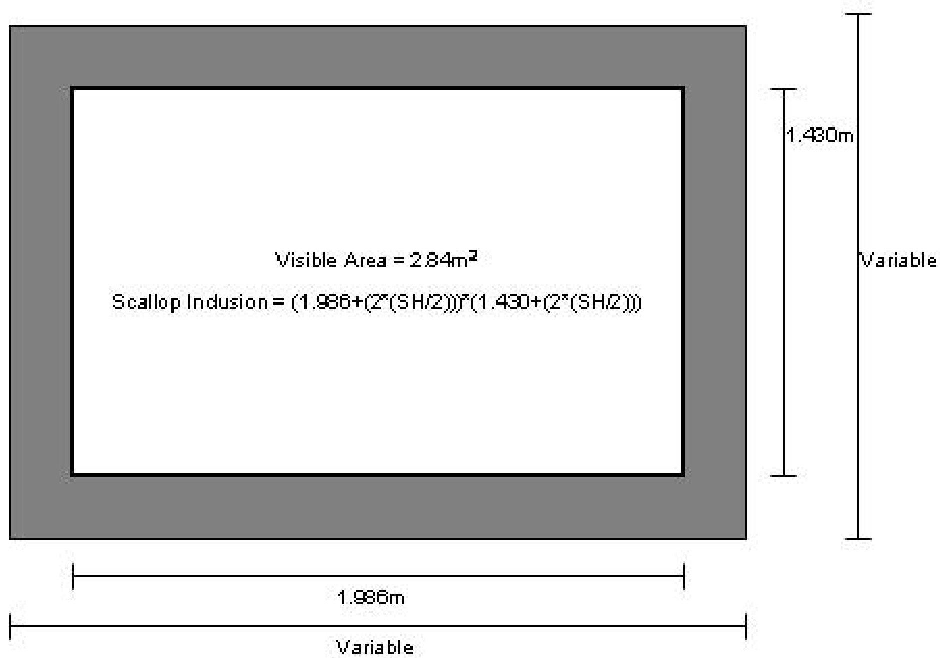
Appendix B3-Table 3. Small camera area surveyed, mean shell height (mm), shell height adjusted view field (m^2), shell height adjusted density, abundance, biomass and 95% CI of the density for the MAB and GBK stock areas from 2003-2009.

SMALL CAMERA

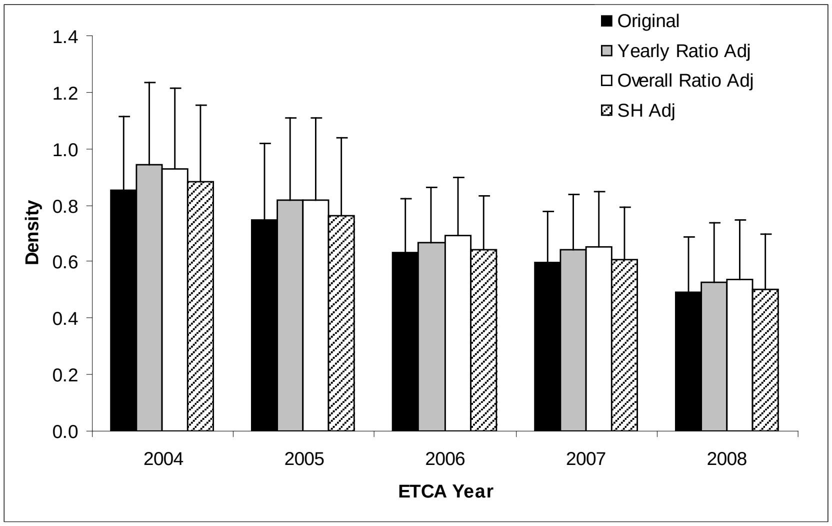
Year	Stations	Area Surveyed km2	Mean SH mm	SH Adj Area m2	SH Adj Density sc/m2	Abundance	Biomass (mt)	95% CI
				Mid-Atlantic Bight				
2003	**799**	**24664**	**58.6**	**0.688**	**0.7063**	**17419973913**	**81353.3**	**0.28**
2004	**829**	**25591**	**84.7**	**0.732**	**0.2319**	**5935561328**	**69251.8**	**0.05**
2005	**860**	**26547**	**87.2**	**0.737**	**0.2181**	**5790580803**	**81756.4**	**0.05**
2006	**872**	**26918**	**93.4**	**0.747**	**0.2049**	**5516773301**	**88322.6**	**0.04**
2007	**931**	**28739**	**90.4**	**0.742**	**0.2204**	**6333997245**	**88940.8**	**0.04**
2008	**913**	**28184**	**90.7**	**0.743**	**0.2160**	**6086579306**	**103164.0**	**0.04**
2009	**928**	**28647**	**98.1**	**0.755**	**0.1260**	**3608213579**	**71935.6**	**0.02**
				Georges Bank				
2003	**904**	**27906**	**88.3**	**0.738**	**0.1698**	**4737032049**	**66669.4**	**0.03**
2004	**921**	**28430**	**101.4**	**0.761**	**0.1256**	**3569624137**	**74431.9**	**0.03**
2005	**902**	**27844**	**111.2**	**0.778**	**0.1001**	**2787348077**	**77928.9**	**0.03**
2006	**916**	**28276**	**109.1**	**0.775**	**0.1412**	**3993072108**	**108804.7**	**0.03**
2007	**901**	**27813**	**80.0**	**0.724**	**0.1974**	**5489504503**	**77728.8**	**0.04**
2008	**882**	**27227**	**99.4**	**0.758**	**0.1526**	**4153894290**	**102841.8**	**0.04**
2009	**942**	**29079**	**96.1**	**0.752**	**0.1556**	**4525694473**	**94067.3**	**0.04**



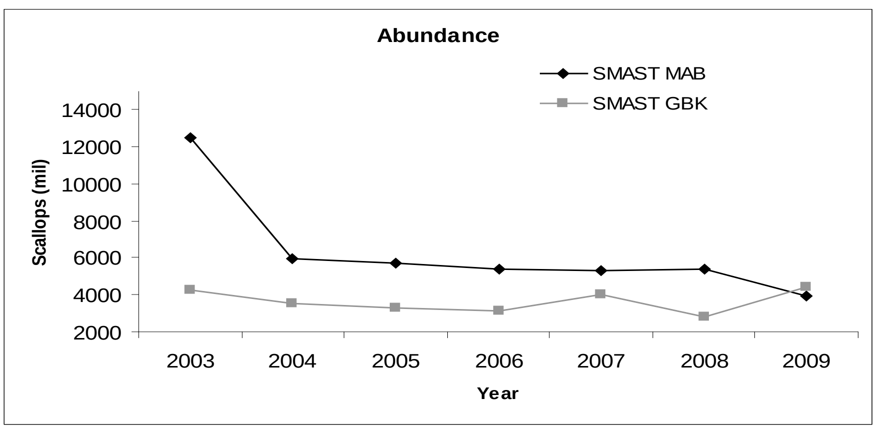
Appendix B3-Figure 1. Camera view field used in calculation of original density.

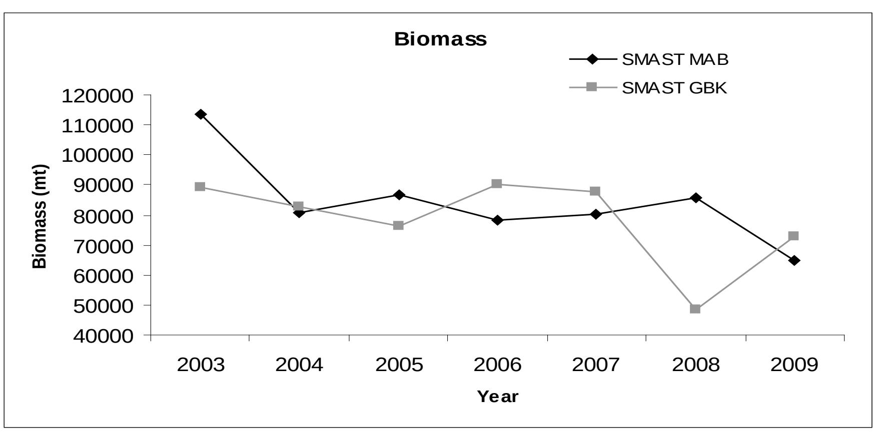


Appendix B3-Figure 2. Camera view field used in calculation of mean shell height adjusted density.



Appendix B3-Figure 3. Density estimates from ETCA 2004-2008 with associated 95% confidence intervals. Data for 2003 and 2009 are not included because the density was much higher (2003) and lower (2009) and muted the 95% CIs for the 2004-2008 data, not allowing comparison.





Appendix B3-Figure 4. Large camera abundance and biomass estimates for MAB and GBK for 2003-2009.

Appendix B4: Vessel calibrations for the NMFS sea scallop survey

In anticipation of the retirement of the R/V *Albatross IV*, the NOAA vessel that had conducted the annual synoptic sea scallop survey virtually uninterrupted since the 1970's, a series of paired tow calibration experiments were conducted to estimate fishing power correction factors. The objective of these experiments was to facilitate the transition of the NMFS sea scallop dredge survey time series from the R/V *Albatross IV* to a future survey platform. Due to some uncertainty in the subsequent survey platform, this information would facilitate the use of the calibrated vessel to either conduct the survey, or at least form a link from the R/V *Albatross IV* to any future survey platform. Ultimately, two calibration experiments were conducted in 2007 and 2009 with the calibration process being conducted in a stepwise fashion. We used a Generalized Linear Mixed Model (GLMM) to analyze the paired catch data to test for differences in both the pooled over length catch data as well as differences in the length composition of the catch. In 2007, the commercial scallop vessel, F/V *Nordic Pride* conducted a paired tow experiment with the R/V *Albatross IV*. Results indicate that while the R/V *Albatross IV* was slightly more efficient, the difference was small (~5%) and not statistically significant. Based on these results, the F/V *Nordic Pride* was considered to be equivalent with respect to fishing power to the R/V *Albatross IV*. In 2008, the R/V *Hugh Sharp* was selected as the replacement vessel for the R/V *Albatross IV* and during the 2009 survey an additional paired tow experiment was conducted between this vessel and the F/V *Nordic Pride*. Results indicate that the R/V *Hugh Sharp* was slightly more efficient (~10%) than the F/V *Nordic Pride*, however, this difference was not statistically significant. These results indicate that scallop dredge catches are robust to the effect of vessel and that any correction factor applied to this time series moving forward is small (~5%) or not justified.

Data collection and analysis

Experimental Design

The calibration experiments were conducted within the context of the NMFS annual sea scallop survey. This survey utilizes a stratified random design to sample throughout the entire U.S. range of the sea scallop. (Serchuk and Wigley 1986). For both paired tow experiments, the sampling occurred during the mid-Atlantic portion of the NMFS survey. For the first experiment, the standard NMFS sea scallop survey dredge that has been in service, virtually unmodified since the 1970's was used aboard both vessels. This dredge is 8 ft in width, with a dredge bag consisting of 2 inch rings. The twine top is comprised of 3.5 inch diamond mesh and there is a 1.5" liner throughout the dredge bag. For the second experiment, the F/V *Nordic Pride* used the standard dredge, while the R/V *Hugh Sharp* used a slightly modified version of the standard dredge referred to as the "prototype'" dredge. The components of the prototype dredge are almost identical to the standard dredge (i.e. ring size, liner mesh size, twine top mesh size). Differences exist in relation to a slightly modified dredge frame, modifications to the ring bag and slight modifications to the mesh counts of the liner and twine top. A major difference between the standard and prototype dredge configurations is the addition of a wheel on the frame of the dredge as well as turtle/rock chains. In essence, the fishing power correction factor estimated for the second experiment attempts to calibrate the existing time series to a new entity that is represented by a unique vessel/gear combination.

While at sea, the sampling protocol included the re-occupation of sampling stations occupied by the R/V *Albatross IV*. Start/stop locations for each tow completed by the R/V *Albatross IV* were relayed to the commercial vessel via VHF radio. With the goal of re-occupying the stations as quickly as possible, a subset of stations was selected for re-sampling (the R/V *Albatross IV* conducts 24 hour operations, while the F/V's in this study sampled for roughly 16-18 hrs/day). During the execution of the tow, the captain of the F/V attempted to mirror the start/stop locations as close as possible. While it is safe to assume that there was some crossing of tow paths, it is unlikely that the tow path was duplicated precisely. For each comparative tow, the dredges were fished for 15 minutes with a towing speed of approximately 3.8-4.0 kts. High-resolution navigational logging equipment was used to accurately determine vessel position and speed over ground. Time stamps from the navigational log in conjunction with the tow level information recorded on the bridge were used to determine the location, duration and area fished by the dredges.

For each paired tow, the entire scallop catch was placed in baskets. A fraction of these baskets will be measured to estimate length frequency for the entire catch. The shell height of each scallop in the sampled fraction will be measured in 5 mm intervals. This protocol allowed for the determination of the size frequency of the entire catch by expanding the catch at each shell height by the fraction of total number of baskets sampled. Finfish and invertebrate bycatch was quantified, with finfish being sorted by species and measured to the nearest 1 cm. Sampling protocol was similar on the R/V *Albatross IV*.

Statistical Models

Scallop catch data from the paired tows provided the information to estimate differences in the fishing power of each vessel/gear combination tested and is based on the analytical approach included in Cadigan *et al.*,2006. Assume that each vessel/gear combination tested in this experiment has a unique catchability. Let q_r equal the catchability of the R/V and q_f equal the catchability of the commercial vessel (F/V *Nordic Pride*) used in the study. The efficiency of the research vessel relative to the commercial vessel will be equivalent to the ratio of the two catchabilities.

$$\rho_l = \frac{q_r}{q_f} \tag{1}$$

The catchabilities of each the vessel/gear combination are not measured directly. However, within the context of the paired design, assuming that spatial heterogeneity in scallop density is minimized, observed differences in scallop catch for each vessel will reflect differences in the catchabilities of the vessel/gear combinations tested. Our analysis of the efficiency of the research vessel relative to the commercial vessels consisted of two levels of examination. The first analysis consisted of an examination of potential differences in the total scallop catch per tow. Subsequent analyses investigate whether scallop size was a significant factor affecting relative efficiency. Each analysis incorporates an approach to account for within-tow variation in the spatial heterogeneity of scallop density.

Let C_{iv} represent the scallop catch at station *i* by vessel *v*, where v=r denotes the research vessel (R/V *Albatross IV* or R/V *Hugh Sharp*) and v=f denotes the commercial vessel (F/V *Nordic Pride*). Let $\bullet_{ir}$ represent the standardized scallop density for the i^{th} station by the R/V and $\bullet_{if}$ the standardized scallop density encountered by the F/V. We assume that due to the tow paths taken by the respective vessels at tow *i*, the densities encountered by the two vessels may vary as a result of small-scale spatial heterogeneity as reflected by the relationship between scallop patch size and coverage by a standardized tow. The standardized unit of effort is a survey tow of 15 minutes at 3.8 kts. which covers a linear distance of approximately .95 nautical miles. The probability that a scallop is captured during a standardized tow is given as q_r and q_f. These probabilities can be different for each vessel, but are expected to be constant across stations. Assuming that capture is a Poisson process with mean equal to variance, then the expected catch by the commercial vessel is given by:

$$E(C_{if}) = q_f \lambda_{if} = \mu_i \qquad (2)$$

The catch by the R/V *Albatross IV* is also a Poisson random variable with:

$$E(C_{ir}) = q_r \lambda_{ir} = \rho \mu_i \exp(\delta_i) \qquad (3)$$

Where $\bullet_i$ =log ($\bullet_{ir}$/ $\bullet_{if}$). For each station, if the standardized density of scallops encountered by both vessels is the same, then $\bullet_i$=0.

If the vessels encounter the same scallop density for a given tow, (i.e. $\bullet_{ir}$= $\bullet_{if}$), then • can be estimated via a Poisson generalized linear model (GLM). This approach, however, can be complicated especially if there are large numbers of stations and scallop lengths (Cadigan *et. al*., 2006). The preferred approach is to use the conditional distribution of the catch by the research vessel at station i, given the total non-zero catch of both vessels at that station. Let c_i represent the observed value of the total catch. The conditional distribution of C_{ir} given C_i=c_i is binomial with:

$$\Pr(C_{ic} = x | C_i = c_i) = \binom{c_i}{x} p^x (1-p)^{r_i - x} \qquad (4)$$

Where p=•/(1+•) is the probability a scallop is captured by the research vessel. In this approach, the only unknown parameters is • and the requirement to estimate µ for each station is eliminated as would be required in the direct GLM approach (equations 2 & 3). For the Binomial distribution *E*(C_{ir})=c_ip and *Var*(C_ir)=c_ip/(1-p). Therefore:

$$\log\left(\frac{p}{1-p}\right) = \log(\rho) = \beta \qquad (5)$$

The model in equation 5, however does not account for spatial heterogeneity in the densities encountered by the two vessels for a given tow. If such heterogeneity does exist then the model becomes:

$$\log\left(\frac{p}{1-p}\right) = \beta + \delta_i \qquad (6)$$

Where $\bullet_i$ is assumed to be normally distributed with a mean=0 and variance=$\bullet^2$. This model represent the formulation to estimate the vessel effect (exp($\bullet_0$)) when scallop catch per tow is pooled over length.

Often, the replacement of a survey vessel presents an opportunity to make changes to the survey fishing gear. In those instances, the potential exists for the catchability of scallops at length, *l* to vary. Even in cases where the survey fishing gear remains the same, length effects are possible. Models to describe length effects are extensions of the models in the previous section to describe the total scallop catch per tow. Again, assuming that between-pair differences in standardized scallop density exist, a binomial logistic regression GLMM model to reflect the situation where one vessel encounters more scallops, but they are of the same length distribution would be:

$$\log\left(\frac{p_i}{1-p_i}\right) = \beta_0 + \delta_i + \beta_1 l, \delta_i \sim N(0,\sigma^2), i = 1,\ldots,n. \quad (7)$$

In this model, the intercept ($\bullet_0$) is allowed to vary randomly with respect to station.

The potential exists, however, that there will be variability in both the number as well as the length distributions of scallops encountered within a tow pair. In this situation, a random effects model that allows both the intercepts ($\bullet_0$) and slopes ($\bullet_1$) to vary randomly between tows is appropriate (Cadigan and Dowden, 2009). This model is given below:

$$\log\left(\frac{p_i}{1-p_i}\right) = \beta_0 + \delta_{i0} + (\beta_1 + \delta_{i1}) l, \delta_{ij} \sim N(0,\sigma_j^2), i = 1,\ldots,n,\ j = 0,1. \quad (8)$$

Adjustments for sub-sampling of the catch and differences in area swept

Additional adjustments to the models were required to account for sub-sampling of the catch as well as differences in the observed area swept by the two gears. In some instances, due to high volume, catches for particular tows were sub-sampled. Often this is accomplished by randomly selecting a subset of the total catch (in baskets) for length frequency analysis. One approach to accounting for this practice is to use the expanded catches. For example, if half of the total catch was measured for length frequency, multiplying the observed catch by two would result in an estimate of the total catch at length for the tow. This approach would artificially overinflate the sample size resulting in an underestimate of the variance, increasing the chances of spurious statistical inference (Millar *et. al.*, 2004; Holst and Revill, 2009). In our experiment, the proportion sub-sampled was consistent throughout each tow and did not vary with respect to scallop length. While experimental protocol dictates a standardized tow of roughly .95 nautical miles (3.8 kts. For 15 minutes), in practice variability exists in the actual tow distances covered by each vessel. These differences must be accounted for in the analysis to ensure that common units of effort are compared.

Let q_{ir} equal the sub-sampling fraction at station i for the vessel r and let d_{ir} be the areal coverage at station i, for vessel r. This adjustment results in a modification to the logistic regression model:

$$\log\left(\frac{p_i}{1+p_i}\right) = \beta_0 + \delta_{i0} + (\beta_1 + \delta_{i1}) l_i + \log\left(\frac{q_{ir} d_{ir}}{q_{if} d_{if}}\right), \delta_{ij} \sim N(0,\sigma_j^2), i = 1,\ldots,n,\ j = 0,1. \quad (9)$$

The last term in the model represents an offset in the logistic regression (Littell, *et. al., 2006*).

In some cases, we encountered difficulties with model convergence for the two parameter model. To simplify the computations in the optimization routine, scallop lengths were standardized to sum 0 based on the interquartile range. This reduced the magnitude of the steps between successive lengths and alleviated the convergence issues. We used SAS/STAT® PROC NLMIXED to fit the generalized linear mixed effects models.

Results and Discussion

Overall, roughly 100 paired tows were completed for each experiment. A visual representation of the spatial distribution of the relative catces for both experiments is shown in Figure 1. For the intercept only model (vessel effect only) a scatterplot of the catches from the paired tows are shown in Figure 2 and parameter estimates are shown in Table 1. For each experiment the R/V was slightly more efficient than the F/V *Nordic Pride* (correction factor is interpreted as $\exp(B_0)$). The calculated correction factors were 1.058 and 1.110 for the two experiments, respectively. In both cases, the logit of the estimated intercept was not significantly different than 0.

For the two parameter model (length effects) there was a significant difference detected in the length composition of catches from the two vessels (Figure 3 and Table 2). The direction of the difference was consistent between the two experiments and showed that the R/V was more efficient as a function of increasing scallop length. The increase in relative efficiency with respect to length for the first cruise may have resulted from measurement errors associated with different measuring devices between the two vessels. For the second experiment, an apparent pattern in the residuals at the small lengths was apparent, however the sum of the animals from lengths <60 mm only represented roughly 4% of the total catch and likely contributed little weight in the likelihood.

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<u>Table 1</u> Mixed effects model (vessel effect only) results including an offset term to account for the effect of differential tow lengths. Parameter estimates are on the logit scale and significant estimates are shown in bold.

Vessel/Gear	$\bullet^{2}$	**Estimate ($\bullet_{0}$)**	**Standard Error**	**Lower 95% CI**	**Upper 95% CI**	**t**	**p-value**
F/V Nordic Pride vs. R/V *Albatross IV*	0.2386	0.0568	0.0501	-0.0427	0.1562	1.13	0.2602
F/V Nordic Pride vs. R/V *Hugh Sharp*	0.4827	0.1040	0.0707	-0.0364	0.2444	1.47	0.1448

<u>Table 2</u> Two parameter mixed effects model results. Both comparisons model the logit of the proportion of the catch at length from the R/V relative to the total catch from both vessels. Parameter estimates reflect a model that includes an offset term in the model that accounted for both sub-sampling of the catch as well as differences in within-tow areal coverage. Confidence limits are Wald type confidence intervals. Parameter estimates are on the logit scale and significant parameter estimates are shown in bold.

Vessel	DF	$\bullet^{2}$ (intercept)	$\bullet^{2}$ (slope)		Estimate	Standard Error	Lower 95% CI	Upper 95% CI	t	p-value
F/V Nordic Pride vs. R/V *Albatross IV*	98	0.2744	0.5077	$\bullet_0$	0.01199	0.05454	-0.09625	0.1202	0.22	0.8264
				$\bullet_1$	**0.4983**	**0.07964**	**0.3402**	**0.6563**	**6.26**	**<0.0001**
F/V Nordic Pride vs. R/V *Hugh Sharp*	98	0.4887	0.3802	$\bullet_0$	0.0908	0.07157	-0.05188	0.2329	1.27	0.2073
				$\bullet_1$	**0.1184**	**0.06879**	**0.05187**	**0.3249**	**2.74**	**0.0073**

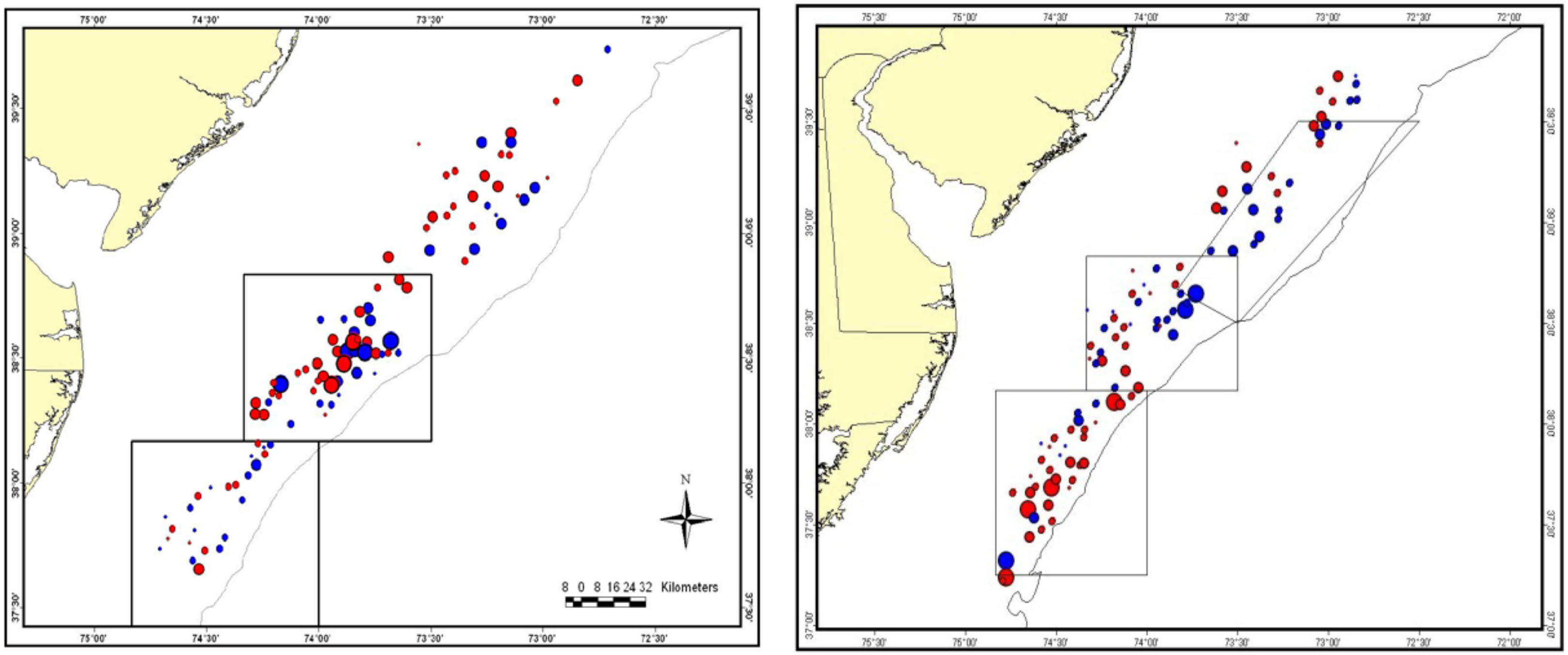
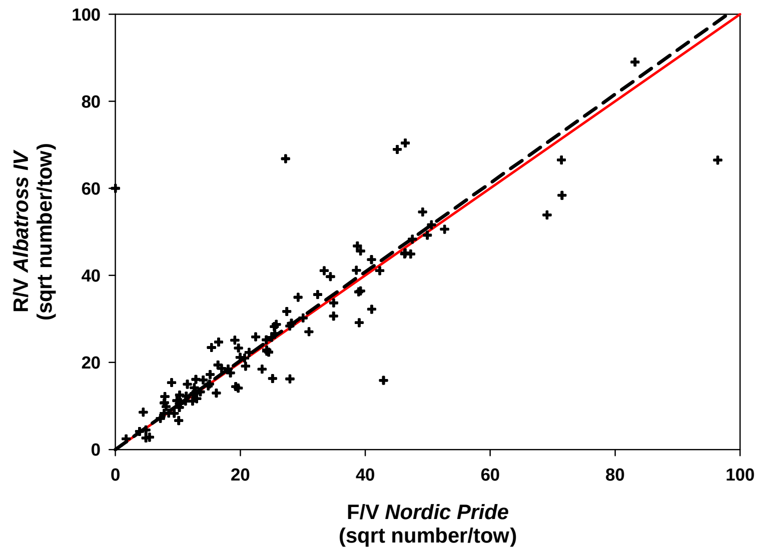
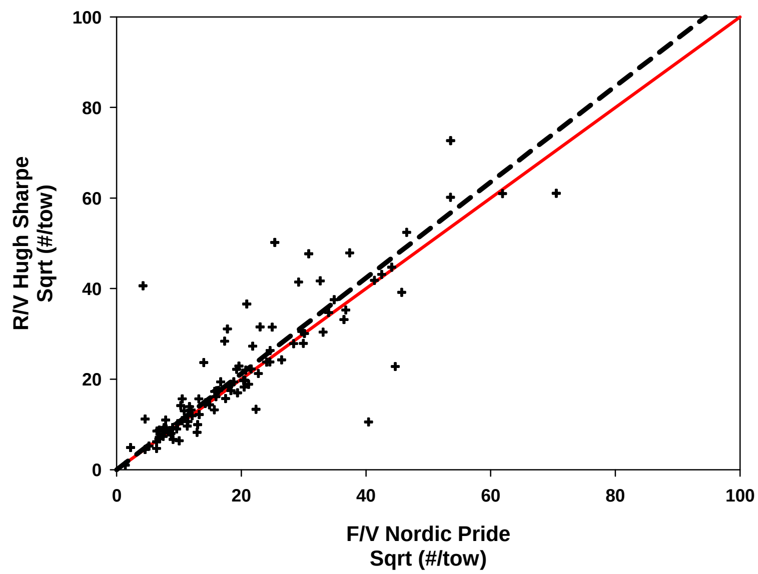


Figure 1 Catch differences between the F/V *Nordic Pride* (towing the standard NMFS dredge) and the R/V *Albatross IV* (left panel) or the R/V *Hugh Sharp* (right panel). Catches for each vessel are scaled to reflect both any sub-sampling of the catch as well as differences in areal coverage. Symbols are proportional to the magnitude of the observed differences in catch. Red dots represent higher levels of catch by the R/V. Blue dots represent higher levels of catch by the F/V *Nordic Pride*. Open circles represent zero difference between the two vessels. Polygons in both areas represent closed areas in existence at the time of the study, which are part of the spatial management strategy for the fishery. The dotted line represents the 50 fathom bathymetric contour.





<u>Figure 2</u> <u>Top Panel:</u> Total scaled catches for R/V *Albatross IV* vs. F/V *Nordic Pride* (top panel) and the R/V *Hugh Sharp* vs. the F/V *Nordic Pride* (bottom panel). The red line has a slope of one. The dashed line has a slope equal to the estimated relative efficiency (from the one parameter vessel effect only model).

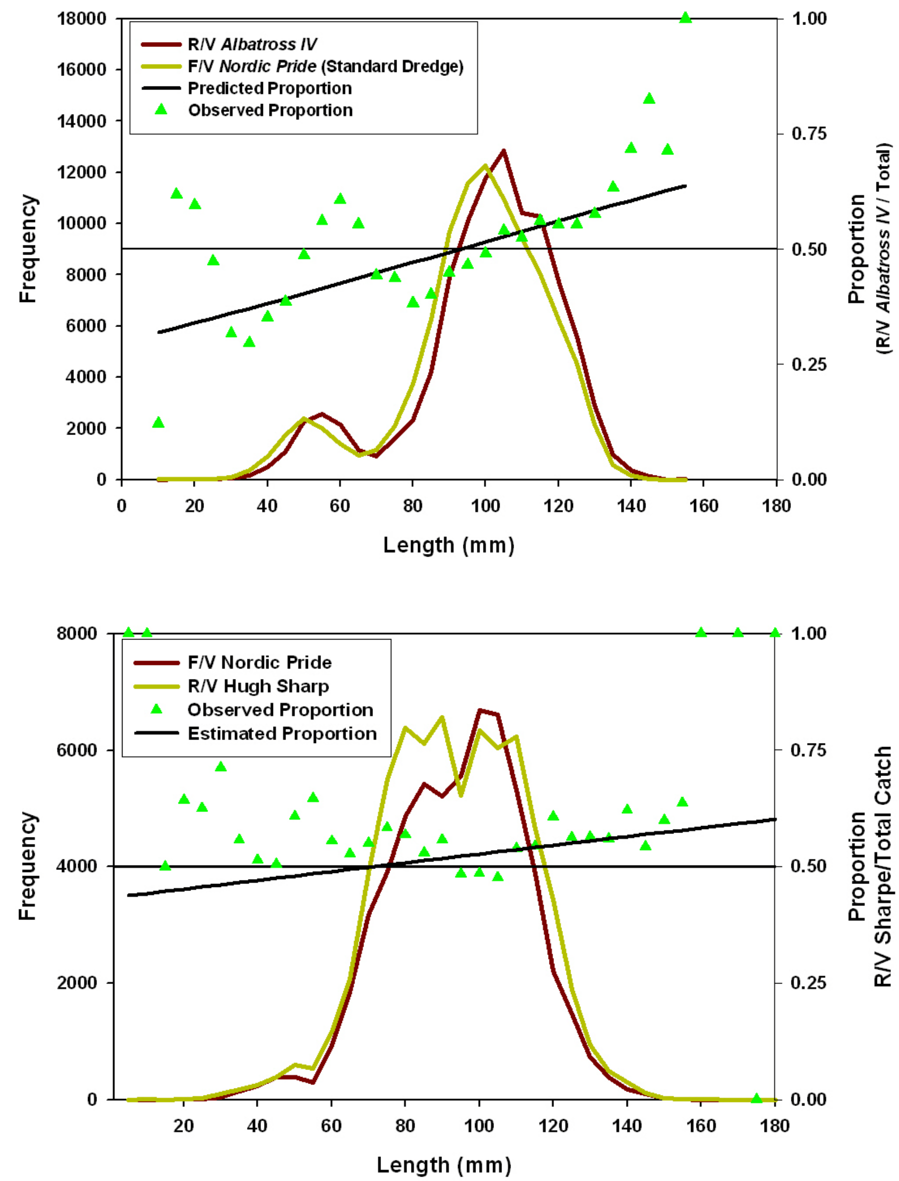


Figure 3 Observed scaled length frequency distributions for the R/V *Albatross IV* and the F/V *Nordic Pride* (top panel) and the R/V *Hugh Sharp* and F/V *Nordic Pride* (bottom panel). The green triangles represent the observed proportions ($Catch_{R/V}/(Catch_{F/V} + Catch_{F/V})$). The black line represents the length based relative efficiency as estimated by the two parameter (vessel and length effect model.

Appendix B5: Results from Maine sea scallop surveys, 2002-2008.

Kevin H. Kelly, Maine Department of Marine Resources, W. Boothbay Harbor, ME.

A dredge-based sea scallop (*Placopecten magellanicus*) survey of Maine state waters (≤ 3 nm from shore) has been conducted since 2002 (with the exception of 2004). This annual survey provides information on size distribution, the shell height-meat weight relationship, abundance, stock size and spatial distribution of scallops from near shore waters along the coast of Maine. For the first two years (2002-2003) the entire Maine coast was surveyed (Schick and Feindel 2005). During 2004-2008, at least one of three major sections of the coast has been surveyed each year on a rotating basis: 1) New Hampshire border to western Penobscot Bay ("Western Maine"); 2) eastern Penobscot Bay to Quoddy Head ("Eastern Maine"); and 3. Cobscook Bay). The following is a chronology of survey coverage by year:

Year	Area surveyed
2002	Coast-wide, including Cobscook Bay
2003	Coast-wide, including Cobscook Bay
2004	no survey
2005	New Hampshire border to western Penobscot Bay
2006	eastern Penobscot Bay to St. Croix River, including Cobscook Bay (*Higher intensity survey than '02 and '03*)
2007	Cobscook Bay
2008	Matinicus Island to Quoddy Head
2009	Cobscook Bay, St. Croix River and New Hampshire border to western Penobscot Bay (*data not yet analyzed*)

The purpose of the survey is to characterize and monitor the sea scallop resource within Maine's coastal waters, and to compare results to previous years' surveys in light of regulatory and environmental changes. It is necessary to monitor changes in abundance and stock size from year to year to evaluate effects of the fishery, document recruitment events and determine what is available for harvest. The survey provides information needed to evaluate potential management strategies such as rotational closures, harvest limits and area closures to protect spawning and enhance recruitment.

Methods

Each survey was conducted aboard a commercial scallop vessel equipped with a standardized survey drag. Vessels were selected by an RFP process where feasible (2005, 2006) but in some cases, particularly in the case of finding a vessel rigged to handle the survey gear and available in the location and time period necessary, there was an additional recruitment process used for vessel procurement.

In some years (2005-2006, and 2008) two vessels were used in order to broaden industry participation, to take advantage of local knowledge and to maximize survey efficiency (the survey was conducted over a broad geographic area with increased sampling intensity and within a fairly narrow time frame). Vessels used were: the *F/V North Star* from Portland (2005); *F/V Sea Ryder* from Spruce Head (2005); 45 ft. *F/V Foxy Lady II* from Stonington (2006, 2008, and

2009); 42 ft. *F/V Alyson J 4* from Cutler (2006, 2008); 40 ft. *F/V Bad Company* from Cutler (2007); and *F/V Kristin Lee* from Eastport (2009).

Surveys were carried out during October-November with the single exception of the fall 2005 survey which was carried over during Feb.-Apr. 2006). Surveys were done during this time to examine scallop size distribution and meat weight in and just prior to the commercial season which starts on December 1 (December 15 in 2009) and to help minimize conflict with lobster traps.

Gear

The survey dredge was a 7 ft. wide New Bedford-style chain sweep with 2½ in. rings in the ring bag to retain smaller scallops (Figure 1). Drag specifications were determined in consultation with several Maine scallop industry members in 2002 prior to the inaugural survey. The dredge was unlined and had rock chains. The twine top was double hung with 3½ in. mesh. The drag size and weight represented a compromise between being wide enough to cover a significant area per tow, heavy enough to sample deeper waters and of a size that can be transported by a large pickup truck (Schick and Feindel 2005).

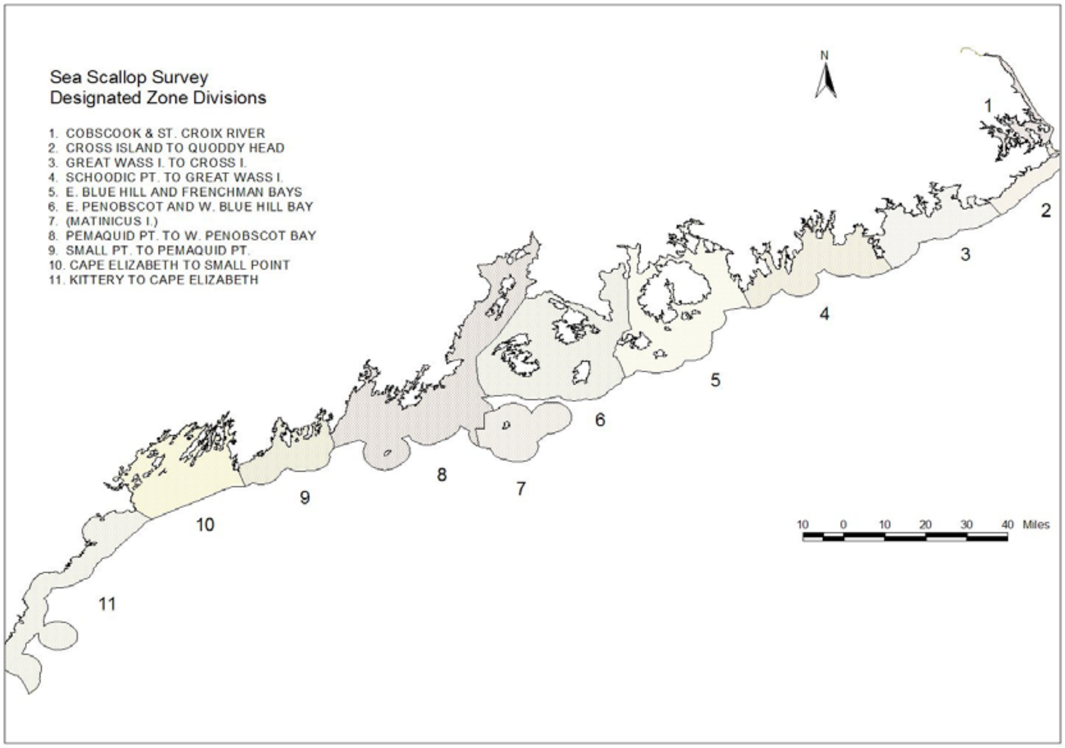
Due to age and wear on the original drags made for the first state waters survey in 2002, survey dredge gear constructed for the 2009 Northern Gulf of Maine in federal waters survey replaced the original gear for the fall 2009 state survey (see Appendix B6). The new gear (Figure 2) was of a configuration largely consistent with that used in previous state surveys but had 2 in. rings to allow better retention of small scallops and a slightly larger pressure plate to facilitate towing in deeper waters.

Appendix B5-Figure 1. View of survey drag used during 2005-2008 showing position of rock chains.

Appendix B5-Figure 2. View of survey drag constructed in 2009.

Survey design

A subset of the coastal zones (or "strata") defined for the 2002-2003 surveys (Figure 3) were used in subsequent surveys during the report period with some modification.



Appendix B5-Figure 3. Survey strata and coastal zones in the Maine DMR scallop survey.

Strata were chosen to provide a manageable balance between area and sampling intensity. Scallop areas within the strata were mapped based on fisher information, prior survey data, sediment maps (http://megisims.state.me.us/metadata/surf.htm) and coastal wildlife inventory maps (http://megisims.state.me.us/metadata/shell.htm) (Schick and Feindel 2005).

Within each stratum except Stratum 1 (Cobscook Bay), survey stations within scallop areas were selected randomly using a 500 m grid (stratified random design). The number of stations assigned within each region was roughly proportional to the size of the strata. There were also a number of fixed stations located in some of the more historically important scallop areas such as Gouldsboro Bay and Libby Islands.

Cobscook Bay is one of 13 survey zones, or "strata", used for the DMR scallop survey. Cobscook Bay is a large, strongly tidal estuary at the extreme eastern edge of the Maine coast near the U.S./Canada border. It has the most productive scallop fishery within Maine waters and is thus sampled with the most frequency and with the highest intensity of the survey zones. A direct assessment of scallop abundance for this stratum is made by using a systematic sampling design.

Six survey substrata (South Bay, Pennamaquan River, East Bay, Whiting Bay, Johnson Bay and area: other) within Cobscook Bay representing spatially contiguous fished areas were determined in consultation with fishing industry members prior to the 2002 survey and have been repeated in subsequent surveys. The total number of stations sampled however was increased by 31% from previous surveys beginning in 2006.

Cobscook Bay tow locations were based on a 500 m grid overlaying each substratum. This grid accommodated an average tow length of approximately 300 m. There were 84 tows completed in the 2007 Cobscook Bay survey and 86 in 2009 (two stations added).

Sampling procedure

Stations to be sampled were plotted using Capn Voyager™ navigational software. A Garmin™ Map 76 GPS unit with Garmin™ GA 29 GPS antenna connected to a laptop computer displaying station location was used to position the vessel on station. Location and time were recorded at three points (dredge in, tow start and haul-back) for each tow. A Juniper Allegro™ ruggedized handheld computer was also connected to a GPS unit to record time/date/location information.

Tow times were 2.5-5 minutes (2.5 minutes in Cobscook Bay) depending on bottom conditions and presence of lobster traps. Stations were sampled by a straight line tow. Boat speed averaged 3.5-4 knots.

A ruggedized handheld computer with an RS232 serial port input for digital calipers was used to facilitate rapid entry of shell measurements and other information while sampling. Data entry screens for the sampling programs and survey were configured using Data Plus Professional™ software, which aided in standardizing data entry, providing error checks and minimizing subsequent data auditing and keying (Schick and Feindel 2005).

The following sampling protocol was employed for each tow:

1) Station information (location, time, depth) was entered from the wheelhouse.
2) Bottom type was recorded as combinations of mud, sand, rock, and gravel based on sounder information and dredge contents. For example “Sg” designated a primarily sand substratum with some gravel (after Kelley et. al.1998).
3) Once the drag was emptied, a digital picture of the haul was taken.
4) Scallops, sea cucumbers (*Cucumaria frondosa*) and ocean quahogs (*Arctica islandica*) were culled from the drag contents for subsequent measurement. Catches of the latter species were quantified because of their importance in other drag fisheries. While the survey gear is not suitable for formally sampling ocean quahogs their presence in the catch does suggest the existence of a bed below the sediment.
5) Bycatch (species other than sea scallops, sea cucumbers and ocean quahogs) was enumerated using a 0-5 qualitative abundance scale corresponding to “absent”, “present”, “rare”, “common”, “abundant”, and “very abundant”.
6) Total number of scallops was recorded. The total weight and volume of the scallop, sea cucumber, and ocean quahog catch was recorded.
7) The shell height (SH; distance from the umbo to the outer edge, perpendicular to the hinge line) of individual scallops was measured. All scallops from catches of 100 animals or less were measured for SH. If >100 scallops were present at least 100 were measured. Where n > 1,000 a subsample of 10% was measured.
8) On selected tows (normally every third or fourth tow) a subsample of 24 scallops, chosen to represent the catch of scallops • 3½ in. shell height, were measured (shell height, shell length and shell depth) and shucked for meat weight determination. Meats were placed in a

compartmentalized box in the order that the animals were measured and later individually weighed on shore (using an Ohaus Navigator™ balance connected to the ruggedized handheld computer) and matched to the corresponding shell measurements.

The following table summarizes data collected for each tow:

Data items collected – ME DMR Sea Scallop survey

COLLECTED DATA - FIELD SUMMARY

TRIP	**STATION INFORMATION** IDENTIFIERS	TOW LOCATION	TOW INFO	ENVIRON. DATA
Trip identifier	Tow identifier	Dredge in (Lat, Lo, Time stamp)	Tow time elapsed	Bottom type
Trip date	Zone	Tow start (Lat, Lo, Time stamp)	Depth	Bottom temperature
Port sailed from	Strata	Haulback (Lat, Lo, Time stamp)	Bearing	
Weather	Location (description)	Drag off-bottom (Lat, Lo, Time stamp)	Wire out	
Precipitation	Tow number	Distance towed	Tow speed	
Wind/ sea stata	Sample type			
Return time	(random, exploratory, "fixed", other)			
Comments				

SCALLOP DATA CATCH	SIZE STRUCTURE	BIOMETRICS	BYCATCH
Number scallops caught	Shell height	Shell height	Tow photo ID
Volume of catch (shellstock)		Shell length	Species
Weight of catch (shellstock)		Shell depth	Abundance (1-5 scale)
Proportion of tow sampled (100, 50, 25%)		Meat weight	Trash type
Number of clappers			Trash amount (1-5 scale)
Coments			Comments

AUXILLARY DATA QUAHOG CATCH	SEA CUCUMBER CATCH	CTD DATA
Number of quahogs	Number of cucumbers	Location (lat/ long)
Shell height	Catch weight	File identifier
Shell length	Catch volume	
Shell depth	Coments	
Shell (dead) abundance (1-5 scale)	Size index (SL x diam 1 x diam 2)	

from Schick and Feindel (2005)

Dredge efficiency

In November 2006, SCUBA transects were conducted in the South Bay substratum of Cobscook Bay in order to compare diver observations of scallop numbers with catch rates of the survey dredge in the same area. At each of three survey stations, five diver transects (covering 2 x 100=200 m^2) were carried out. All scallops in each dive transect were measured for shell height and counted. These stations (SM1S39, SM1S46, and SM1S51) were located in areas of higher scallop density in South Bay. At each station two (2) replicate tows from each of the two (2) survey vessels (n = 4) were also performed to determine size-specific scallop density by dredge for comparison.

The diver transects indicated that the survey drag was 43.6% efficient at capturing scallops $\geq$ 95.25 mm (3 ¾ in) SH. (This shell height was chosen as it represented the minimum legal size of scallops in Maine in 2003 and dredge efficiency is of particular importance for estimating harvestable (minimum legal size and above) biomass. This efficiency estimate is less than previously reported for the survey dredge (68.0%; Schick and Feindel 2005) but compares favorably with the efficiency estimate for the NMFS survey dredge (45% in Closed Areas I and II on Georges Bank; NMFS/NEFSC 2004). Our estimate also compares well with efficiency estimates from other New England-style commercial dredges (42.7%; Gedamke et al. 2004). For the cooperative survey of scallop abundance in Closed Area II using commercial-type gear (SMAST, VIMS, Fisheries Survival Fund, NMFS), commercial dredge efficiency was estimated to be 53.1 – 54.4% (Gedamke et al. 2005). The DMR dredge is unlined and therefore would be expected to have higher efficiency for legal scallops than a lined dredge (D. Hart, NMFS/NEFSC, pers. comm.). The particular bottom type of our dredge efficiency study sites was largely sandy gravel, typical of much of Cobscook Bay, which also likely increased gear efficiency compared to more rocky areas along much of the rest of coastal Maine. Given these considerations, the estimate of 43.6% efficiency is plausible.

Data analysis

Area swept per tow was determined from tow distance (tow start to haul-back) and drag width (7 ft. or 2.1 m). Tow distance was determined using Capn Voyager™ software. Based on this information, the scallop catch for each tow was standardized to density (number of scallops per square meter). Total abundance was calculated by multiplying density and area.

For analysis, total scallop catch was divided into the following size categories:

- “seed”: < 2½ in. (<63.5 mm) SH
- “sublegal”: 2½ in. to < 4 in. (63.5 – <101.6 mm) SH
- “harvestable”: • 4 in. (• 101.6 mm) SH

Estimates of total abundance for each of the three size classes were calculated using Cochran’s (1977) standard approach for surveys. For each of the six survey substrata identified above, the average density was estimated as:

$$\bar{X} = \sum_{h=1}^{H} W_h \bar{X}_h$$

where $\bar{X}_h$ the average density for substratum h, H is is the total number of substrata, and W_h is proportion of the area of substratum *h* with respect to the survey area. The associated standard error was calculated:

$$std\ error(\bar{X}) = \sqrt{\sum_{h=1}^{H} W_h^2 \frac{1-f_h}{n_h} S_h^2}$$

where S_h^2 is the variance estimated for substratum *h*, $f_h = \frac{n_h}{N_h}$ is the finite population correction for substratum *h*, and n_h and N are the number of stations sampled and the total number of stations available for sampling, respectively, in substratum *h*. The finite population correction factor was ignored since the proportion of area sampled was small compared to the total area of each substratum.

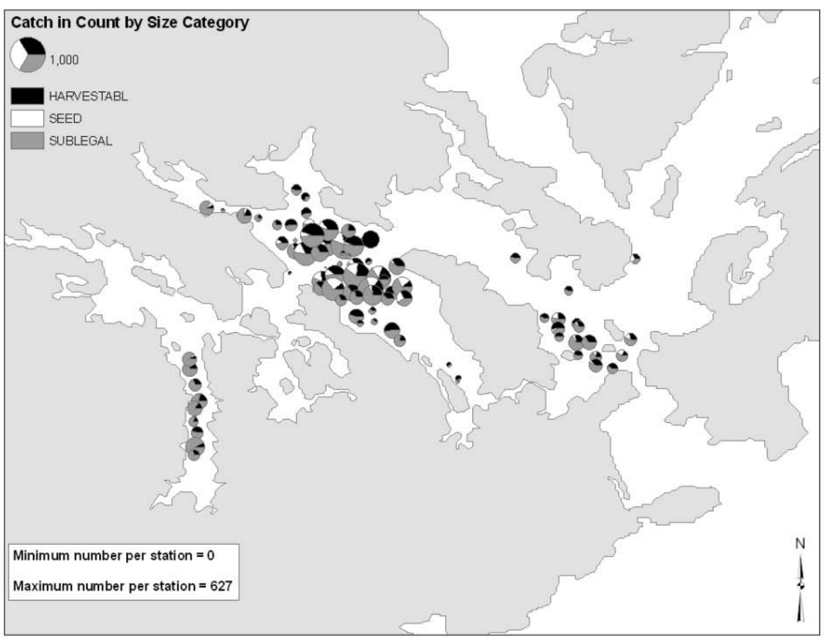
Harvestable biomass for Cobscook Bay was calculated by applying a calculated shell height-meat weight relationship to the numbers of harvestable scallops at shell height per substratum. Biomass was summed across substrata to determine total harvestable biomass for Cobscook Bay.

Results

Cobscook Bay was surveyed in 2003, 2006 and 2007. The survey indicated a large increase in abundance and biomass of harvestable (• 4 in. SH) scallops in Cobscook Bay between 2006 and 2007.

The abundance of harvestable scallops in 2007 was 96.2% greater than the previous high observed in 2003. This increase appears plausible because it followed the high abundance of sublegal (2.5 – 3.9 in. SH) scallops observed in 2006.

Although sublegal scallop abundance declined in 2007 from the high level of 2006 the density of seed (< 2.5 in. SH) was significantly (p=0.008) higher in South Bay in 2007 (0.064 m^{-2}) than 2006 (0.025 m^{-2}) (Table 1; Figure 8). Recruitment, although not as high as in 2006, appeared healthy in 2007 as considerable numbers of both seed and sublegal scallops were present in South Bay, the largest and most important fishing ground (Table 1; Figures 4-11).

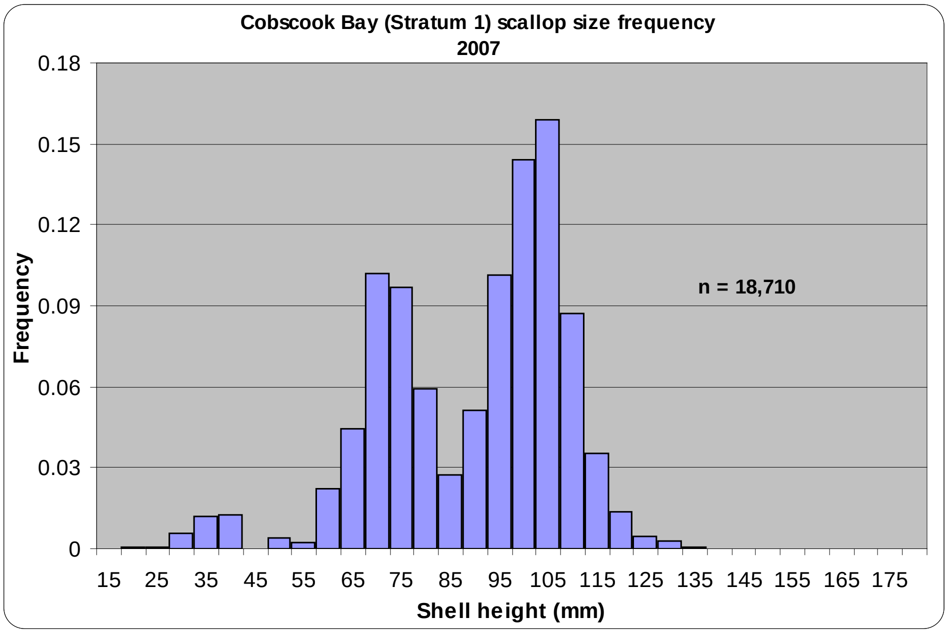


Appendix B5-Figure 4. Scallop size class composition and abundance (Cobscook Bay), *2007 survey.*

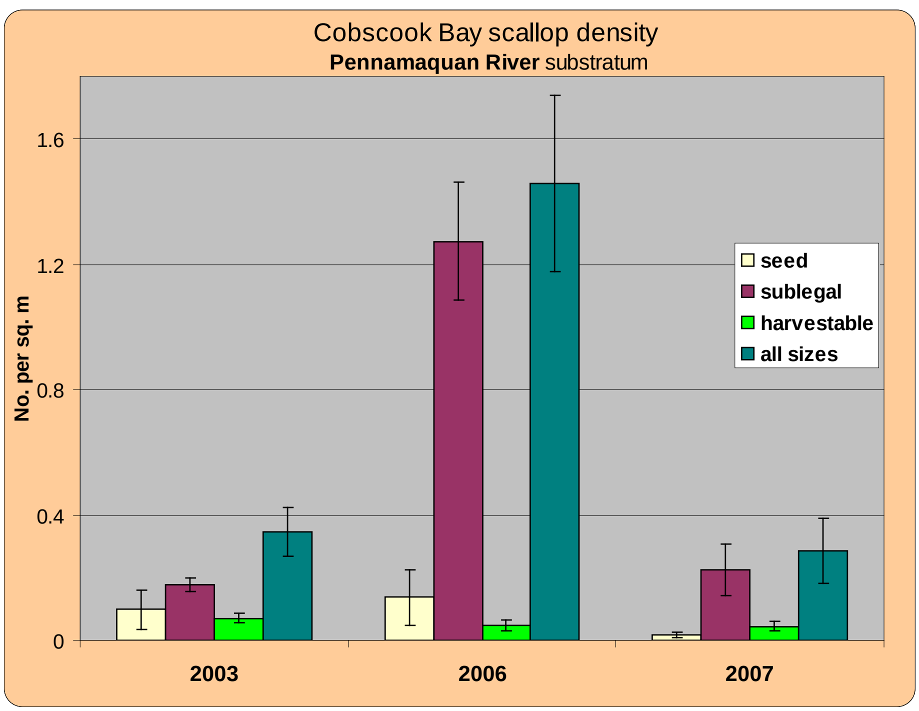
Appendix B5-Table 1. Survey summary statistics for Cobscook Bay (2007) by substratum and overall (mean +/- standard error).

Stratum 1 (Cobscook Bay) scallop survey - 2007

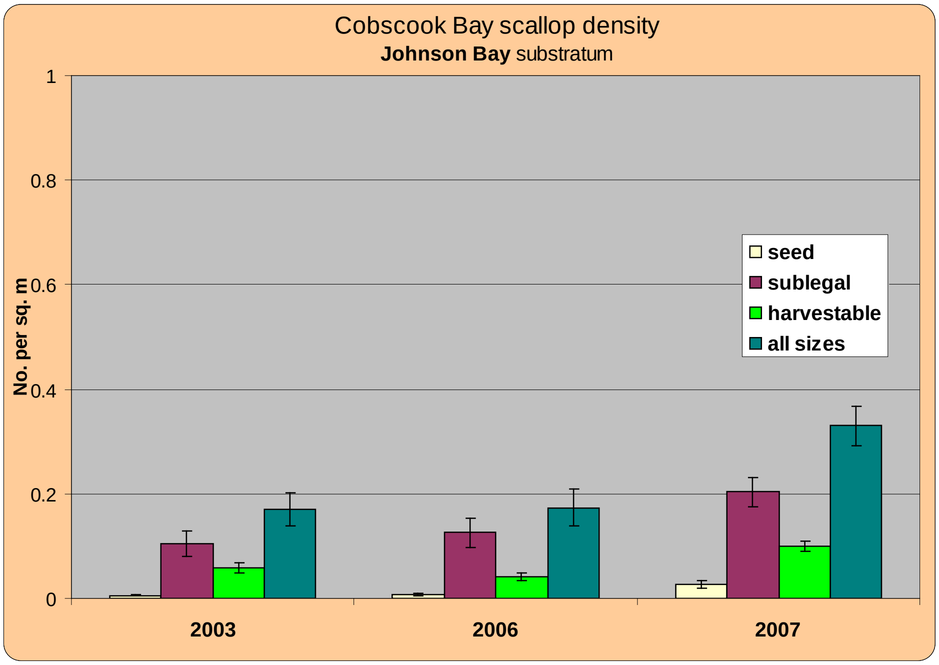
substratum	**total**		**South Bay**		**East Bay**		**Penn. River**		**Whiting Bay**		**Johnson Bay**		**other**	
area (hec)	2,158		1,182		92		64		135		401		284	
no. sites	83		48		3		5		9		15		3	
			Density (scallops per sq m)											
			density	S.E	density	S.E	density	S.E	density	S.E	density	S.E	density	S.E
seed			0.064	0.013	0	0	0.017	0.009	0.004	0.002	0.027	0.006	0.029	0.027
sublegal			0.345	0.042	0.108	0.031	0.225	0.083	0.338	0.062	0.203	0.028	0.107	0.011
harvestable			0.147	0.018	0.144	0.008	0.045	0.017	0.060	0.009	0.099	0.010	0.089	0.010
all sizes			0.556	0.066	0.252	0.037	0.287	0.103	0.402	0.063	0.330	0.038	0.224	0.037
			Abundance (no. scallops)											
	abundance		abundance	S.E	abundance	S.E	abundance	S.E	abundance	S.E	abundance	S.E	abundance	S.E
seed	964,714		757,544	147,935	0	0	10,792	5,531	5,655	2,487	108,018	25,975	82,706	76,000
sublegal	5,891,034		4,073,386	500,090	99,133	28,358	143,899	53,111	455,899	83,118	815,680	111,276	303,037	31,850
harvestable	2,635,277		1,741,962	210,599	132,439	7,599	28,885	10,665	81,462	12,449	398,798	39,610	251,731	27,170
all sizes	9,491,025		6,572,892	785,229	231,572	33,669	183,576	66,200	543,016	84,968	1,322,495	153,474	637,474	105,264
	Harvestable biomass (kg) (unadjusted)													
	biomass	S.E	biomass	S.E	biomass	S.E	biomass	S.E	biomass	S.E	biomass	S.E	biomass	S.E
	55,637	6,712	36,084	4,444	2,921	128	560	202	1,620	256	8,757	857	5,696	825



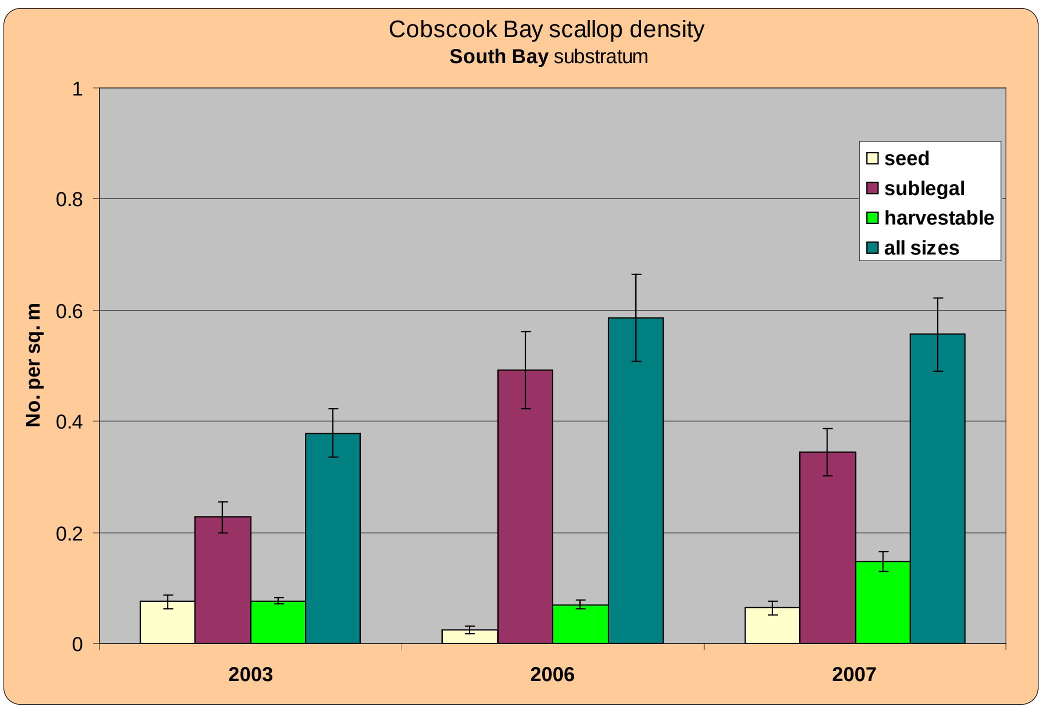
Appendix B5-Figure 5. Size frequency (5 mm increments) of scallops in Cobscook Bay, 2007.



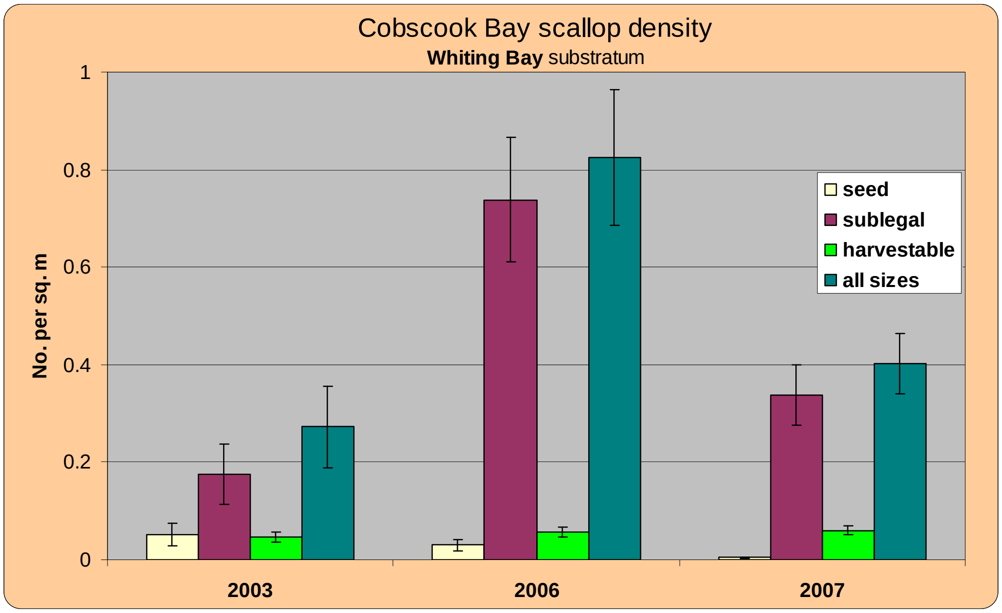
Appendix B5-Figure 6. Mean scallop density (+/- one standard error, unadjusted for dredge efficiency) by size class, Pennamaquan River substratum of Cobscook Bay.



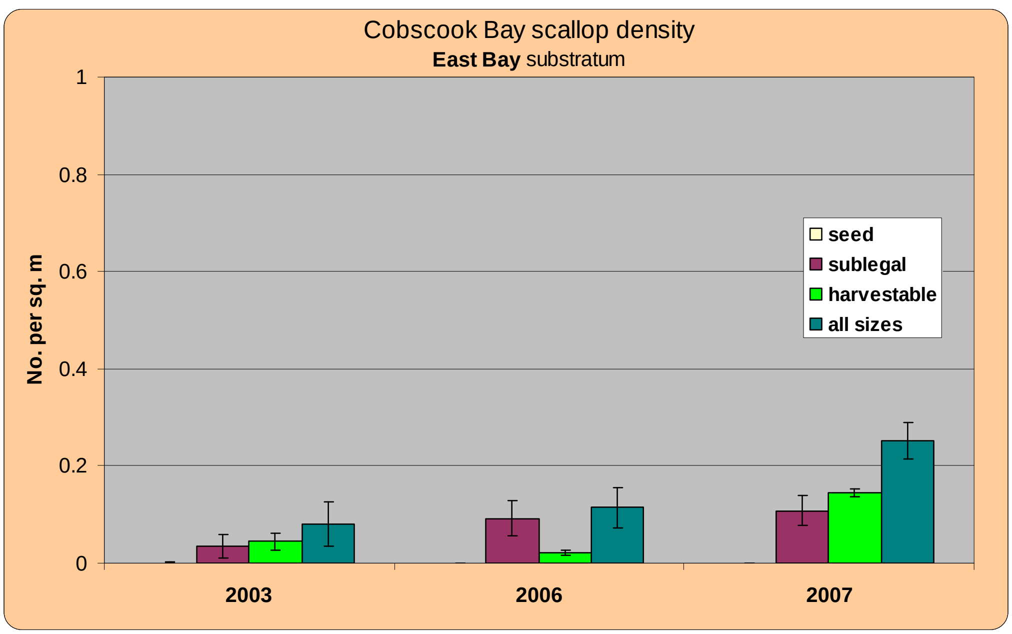
Appendix B5-Figure 7. Mean scallop density (+/- one standard error, unadjusted for dredge efficiency) by size class, Johnson Bay substratum of Cobscook Bay.



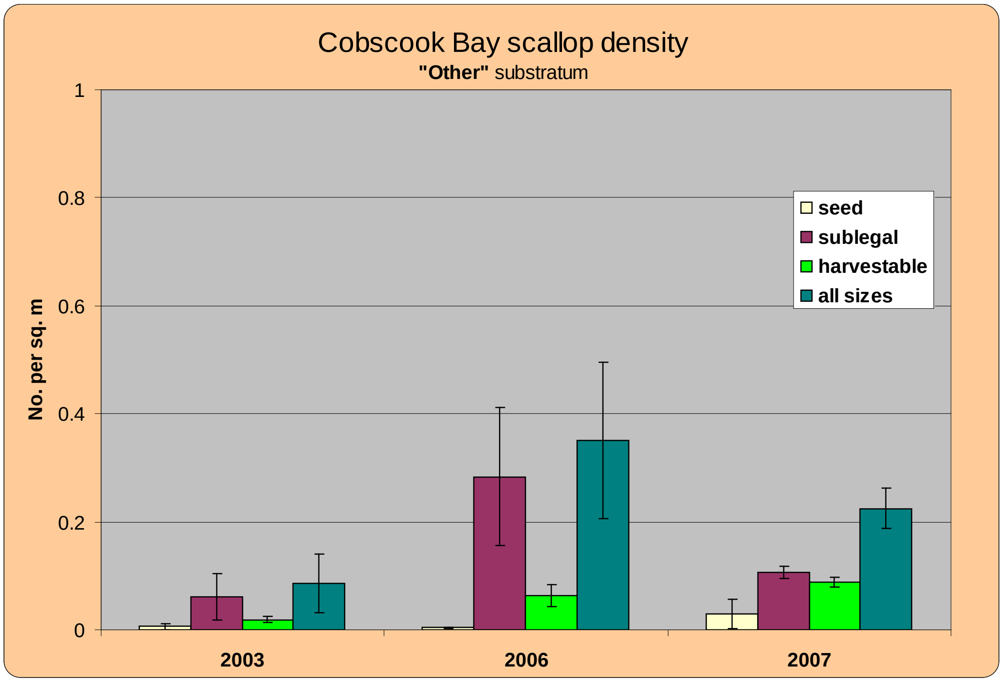
Appendix B5-Figure 8. Mean scallop density (+/- one standard error, unadjusted for dredge efficiency) by size class, South Bay substratum of Cobscook Bay.



Appendix B5-Figure 9. Mean scallop density (+/- 1 standard error, unadjusted for dredge efficiency) by size class, Whiting Bay substratum of Cobscook Bay.



Appendix B5-Figure 10. Mean scallop density (+/- one standard error, unadjusted for dredge efficiency) by size class, East Bay substratum of Cobscook Bay.

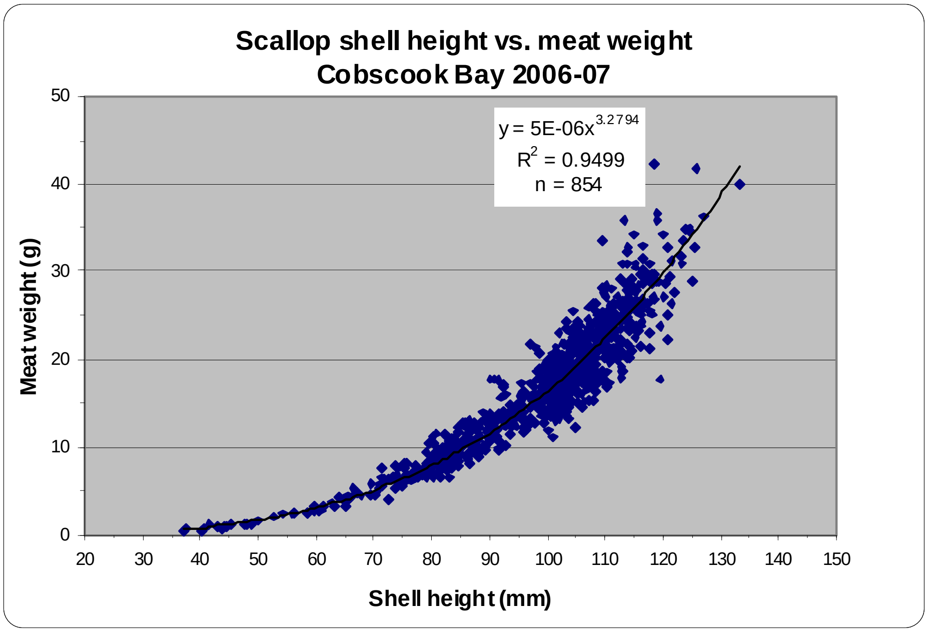


Appendix B5-Figure 11. Mean scallop density (with standard error, unadjusted for dredge efficiency) by size class, "other" substratum of Cobscook Bay.

Shell height-meat weight

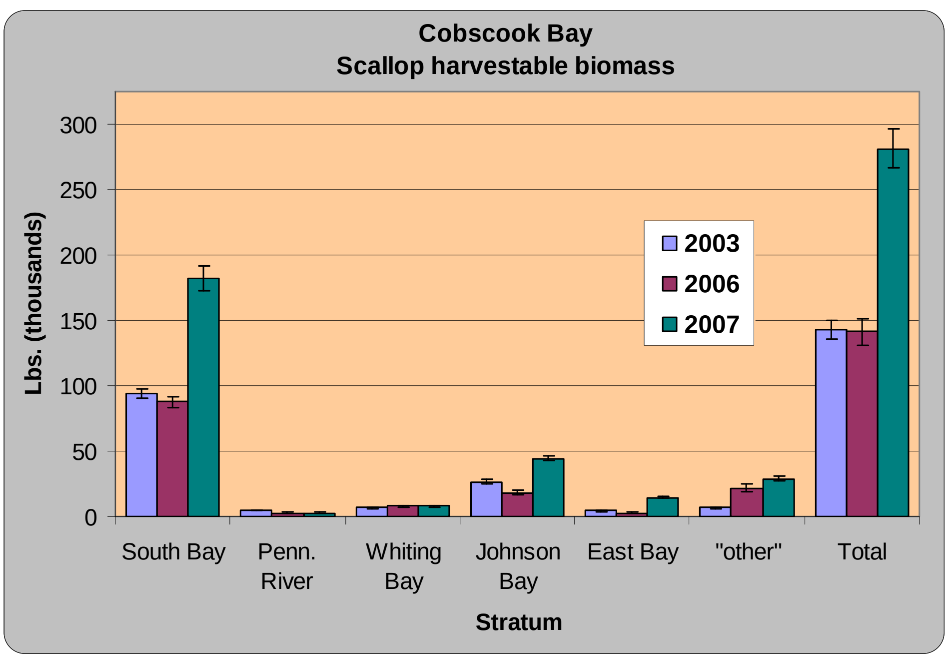
A shell height to meat weight relationship was calculated based on samples taken in 2006-2007 (Figure 12). Scallop meat weights from 2006-2007 were lower than those in 2002-2003 (18% less meat weight at 4 in. SH). The 2006-2007 relationship ($MW = 0.00000453\ SH^{3.2794}$) differed significantly from the 2002-2003 equation ($MW = 0.000037\ SH^{3.365}$) for Cobscook Bay (Schick and Feindel 2005).

Meat weights were greater in 2002-2003 than in 2006-2007. The 2006-2007 meat weights were larger however than those reported for 1987 and 1991 in an unpublished DMR study where the relation was $MW = 0.000005\ SH^{3.2247}$. It should be noted that the 1987 and 1991 studies were based mainly on smaller (80-100 mm) scallops than those sampled in the more recent surveys (minimum legal size was 3.0 in. or 76.2 mm) until 1999). Thus predicted meat weights for scallops in the current legal size range (• 4 in.) from the 1987/1991 report may be less reliable than the more recent studies. Furthermore the 1987 and 1991 sample sizes were relatively small (n = 296). The 1987 and 1991 studies do provide some evidence that the 2006-2007 data are within a "normal" range for Cobscook Bay and still higher than overall meat weights for coast-wide Maine (Schick and Feindel 2005). The 2006-2007 commercial meat counts (26 per lb. at the 4 in. SH minimum size) also appeared well below the legal maximum commercial meat count (35 per lb.) for Cobscook Bay.



Appendix B5-Figure 12. Scallop meat weight (MW) as a function of shell height (SH) for Cobscook Bay, 2006-2007.

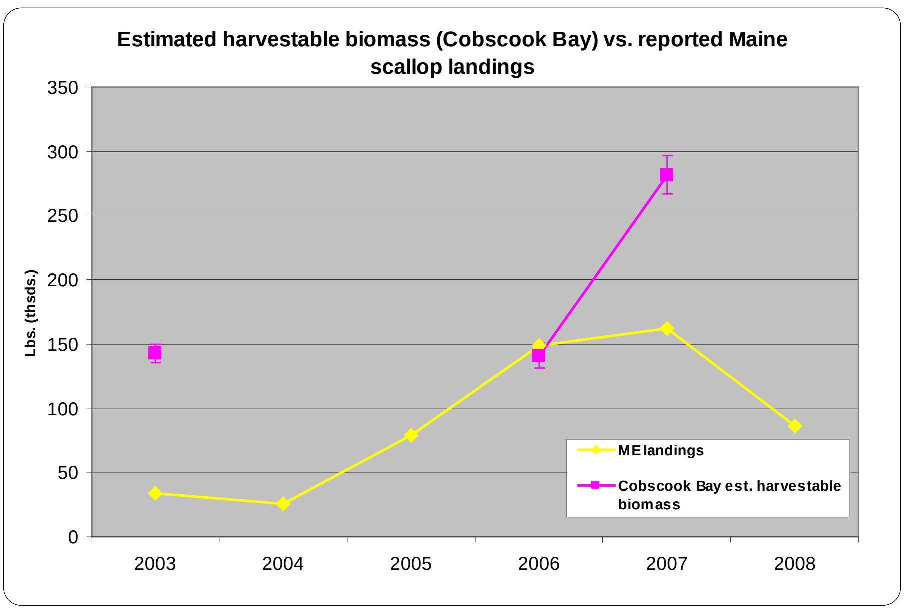
The 2007 estimate of harvestable biomass (128 mt or 281.3 thousand lbs of meats) was 99.4% higher than the previous year (Figure 13). South Bay had the largest proportion (65%) of harvestable biomass.



Appendix B5-Figure 13. Biomass (meat weight) of harvestable (legal-size) scallops in Cobscook Bay in 2003, 2006 and 2007.

An economic study (Athearn 2005) indicated that Cobscook Bay landings for the 2004-2005 season were 70.3 mt (155 thousand lbs) or meats. However, landings data for calculation of exploitation rates in Cobscook Bay were generally not available for years with surveys. Scallop harvesters in Maine were been required to report trip level information, including landings, beginning with the 2008-2009 season but there is too little information available from which to determine Cobscook Bay scallop landings for earlier years. Maine landings prior to 2008 were determined by a voluntary dealer reporting system which did not provide information on where the scallops were caught. Furthermore, many Cobscook Bay harvesters have traditionally "peddled" or retailed their scallops directly to consumers rather than sell to a dealer.

Based on industry input, observations from port sampling, the amount of resource available as observed on the dredge survey and the high level of fishing activity there, that a very large portion (perhaps 80-90%) of overall Maine scallop landings are from Cobscook Bay. A comparison of estimated harvestable biomass (Cobscook Bay) and reported Maine landings does not, however, show a high correlation (Figure 14), except for the slight trend upward in 2007 landings concurrent with the large increase in Cobscook Bay biomass. It is hoped that improved comparisons can be made beginning when 2009 survey data become available along with 2009-10 harvester reports.



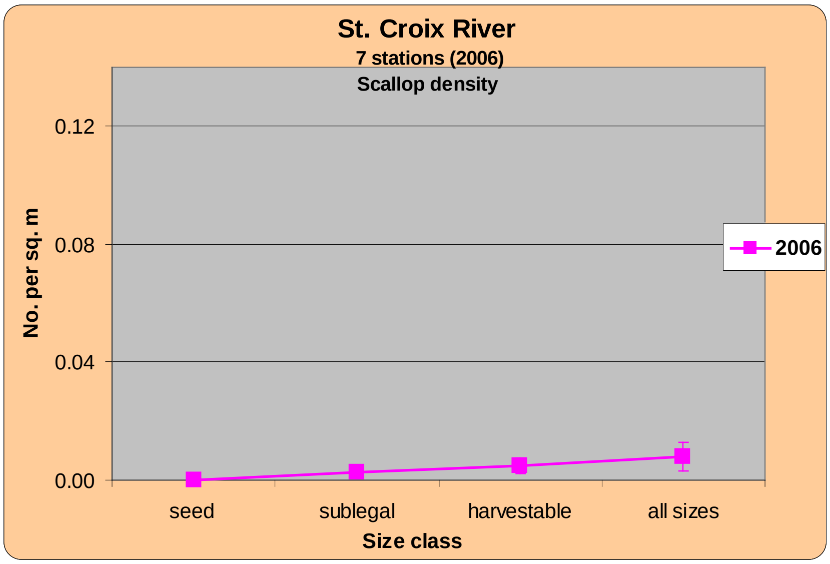
Appendix B5-Figure 14. Cobscook Bay harvestable biomass as estimated by DMR survey in relation to reported Maine scallop landings, 2003-2008.

Cobscook Bay continued to exhibit relatively high scallop production during 2003-2008 despite the intense fishing effort which existed there. There are no official reports of fishing activity but it has been stated for example that 170 boats were operating there on opening day 1995 (Cobscook Bay Resource Center 2007). Maine Marine Patrol estimated that 90-100 vessels were fishing in Cobscook Bay in mid-December 2007 (Lt. A. Talbot, pers. comm.).

On the 2009 survey of Cobscook Bay/St. Croix River, approximately 20,400 scallops were caught and counted, 8,700 were measured for shell height and an additional 800 were sampled for shell size-meat weight determination. The new dredge with 2" rings seemed particularly effective at sampling across the full size range of the resource. Data analysis and a report for this survey will be completed in 2010.

Stratum 1a (St. Croix River)

The St. Croix River was surveyed in 2002 and 2006. This stratum was characterized by relatively low scallop abundance (0.005 m^{-2}) in 2006 with harvestable sizes (0.003 m^{-2}) slightly more abundant than sublegals (0.002 m^{-2}) (Figure 15). Catch rates were also low in 2002 (Schick and Feindel 2005). The highest survey catch rate in 2006 was around Frost Island near Passamaquoddy Bay.



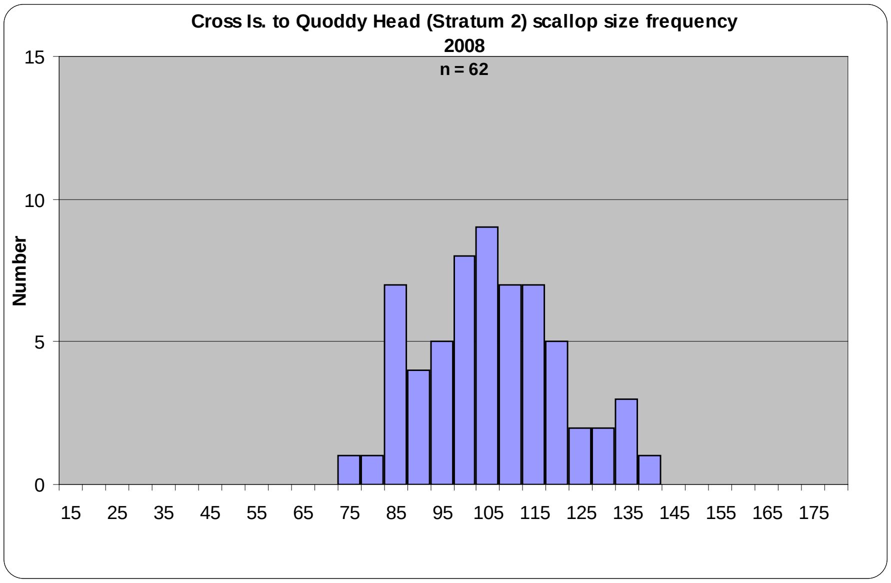
Appendix B5-Figure 15. Mean scallop density by size group, Stratum 1A.

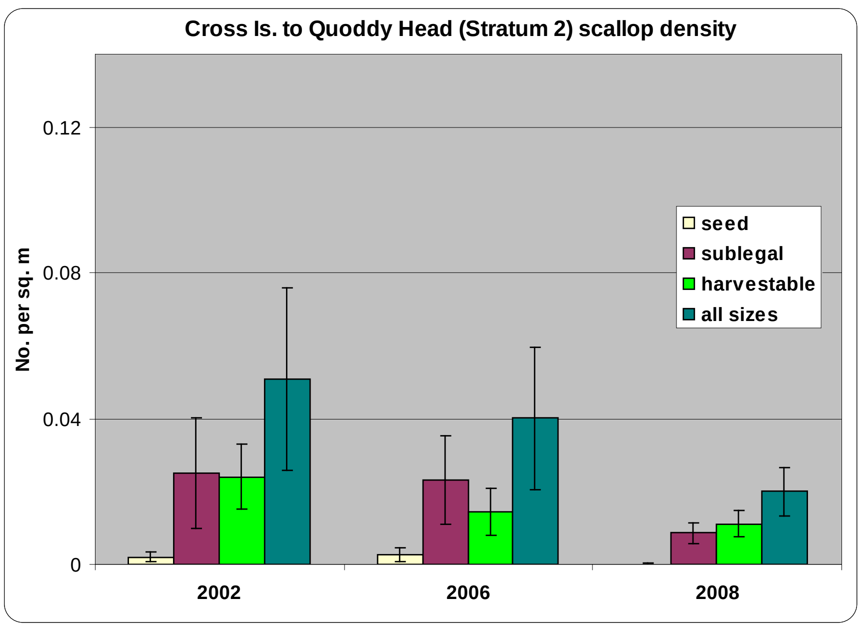
Eastern Maine: Strata 2-7 (Quoddy Head to Matinicus Island)

These strata were surveyed in 2005 (Stratum 7), 2006 (Strata 2-6) and 2008 (Strata 2-7). There were 183 tows completed in 2008. Most of the tow locations were randomly selected within the known scallop grounds of each stratum. The survey indicated that overall scallop abundance either declined slightly or remained unchanged at a low level of abundance for all areas except Stratum 6 (East Penobscot Bay and W. Blue Hill Bay). A slight increase was observed in the latter area (Figure 20). Although densities remained fairly low in this stratum, the size distribution indicated some successful recruitment.

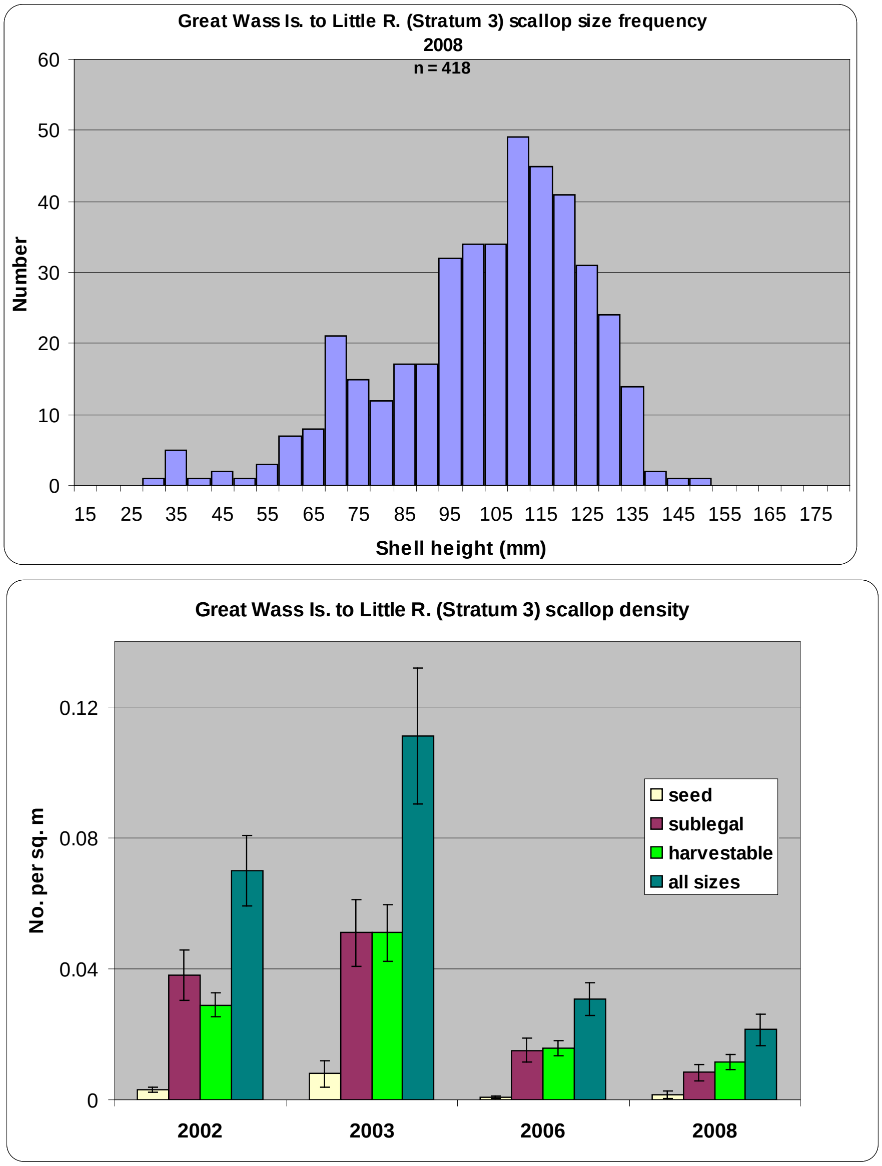
Considerably higher densities had been observed in Stratum 3 (Great Wass Island to Little River), an area of relatively high fishing pressure. Densities were 0.111 m^{-2} in 2003, 0.031 m^{-2} during 2006 and 0.021 m^{-2} during 2008 (Figure 17). The size range in this stratum has shifted to older, larger scallops (similar to Stratum 4 in 2006) indicating reduced recruitment.

The presence of seed scallops (< 2½ in. shell height) was noted at six (6) locations in the overall eastern Maine area in 2008.

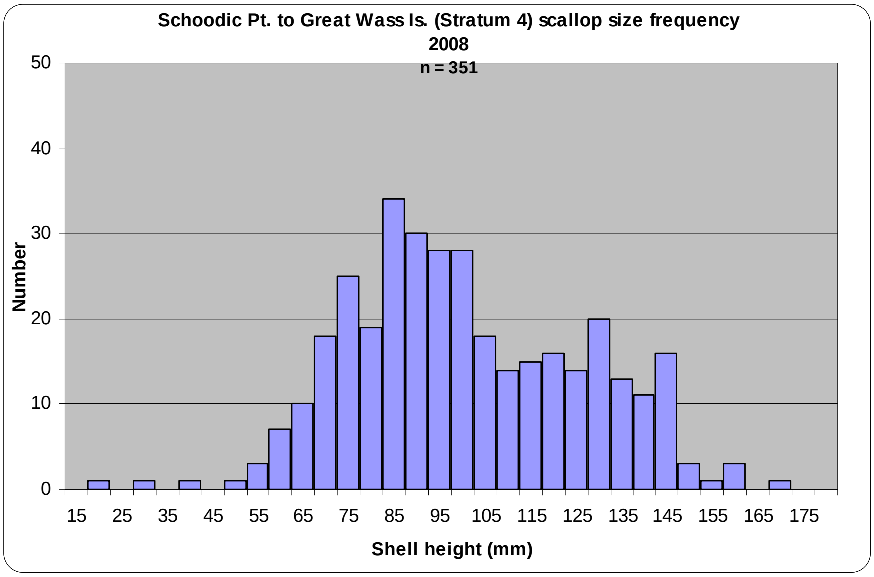


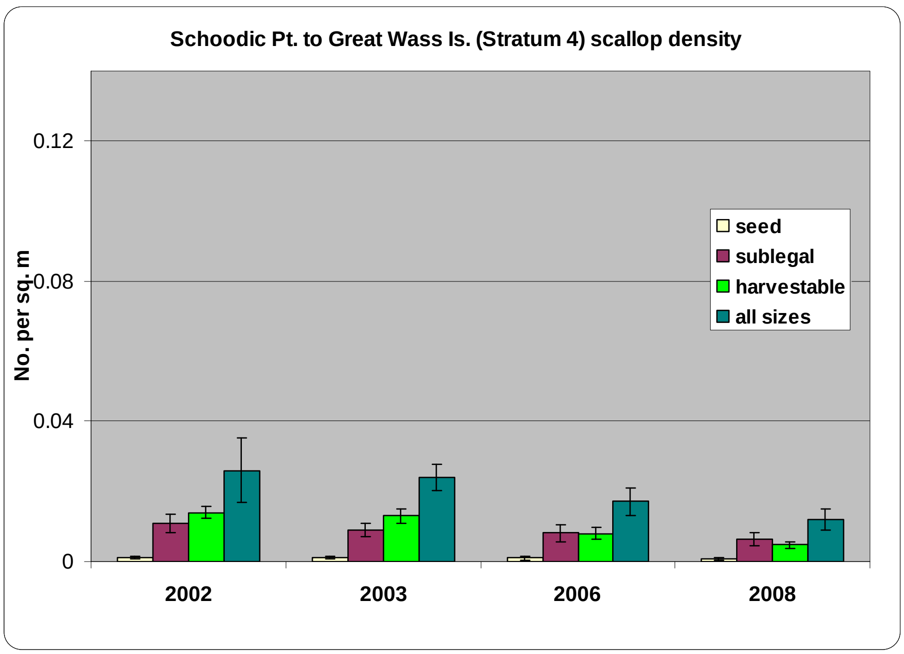


Appendix B5-Figure 16. Scallop size frequency (5 mm increments) (top) and mean density (+/- one standard error, unadjusted for dredge efficiency) by size class (bottom), Cross Island to Quoddy Head.

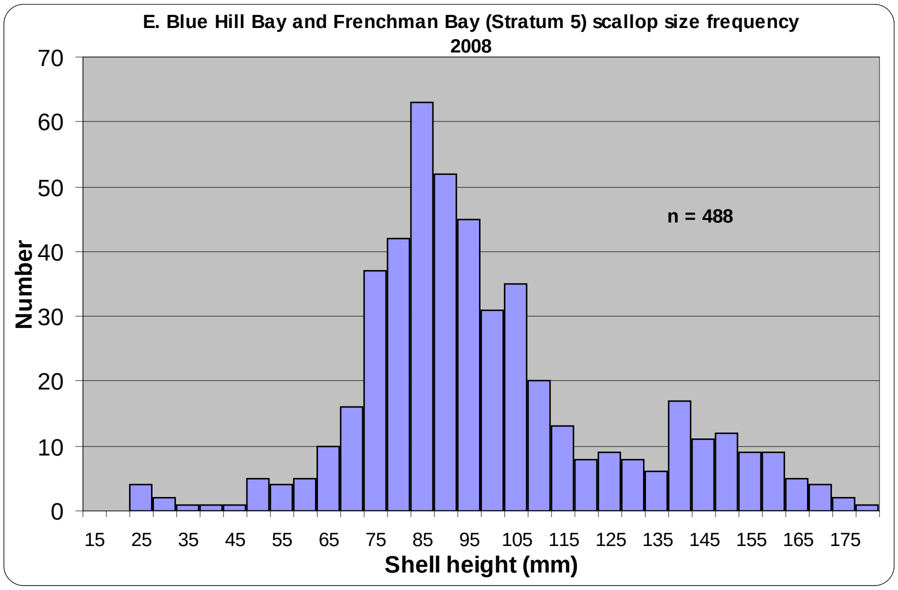


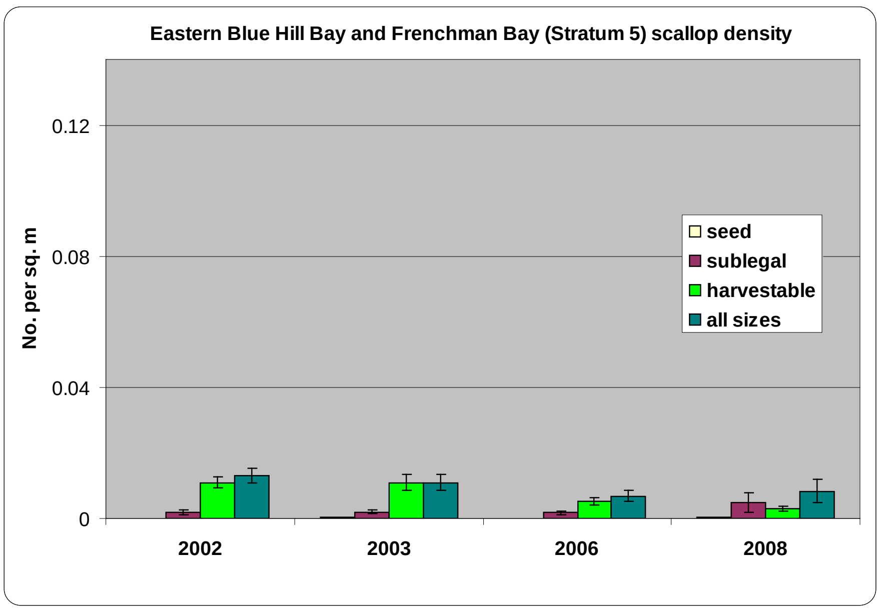
Appendix B5-Figure 17. Scallop size frequency (5 mm increments) (top) and mean density (+/- one standard error, unadjusted for dredge efficiency) by size class (bottom), Great Wass Island to Little River.



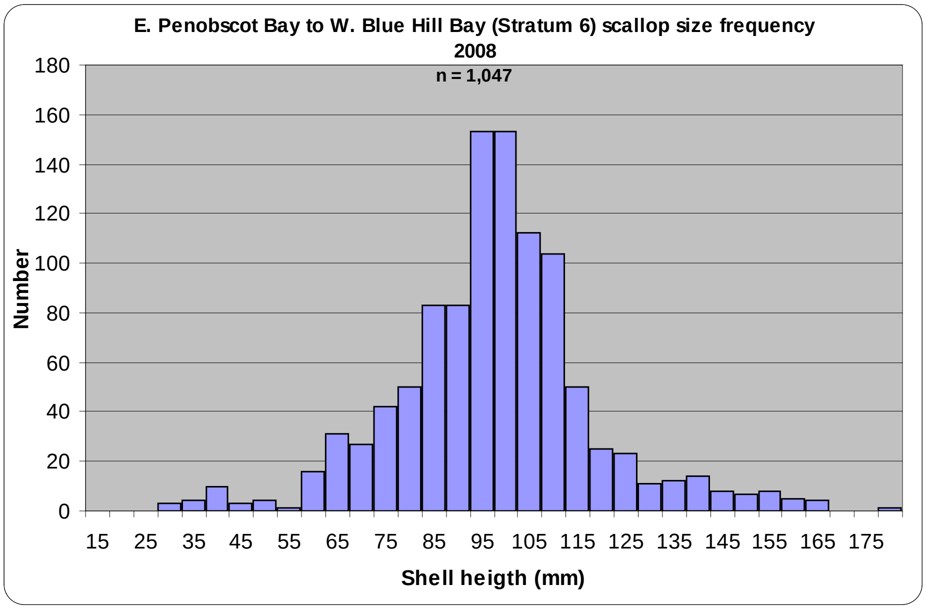


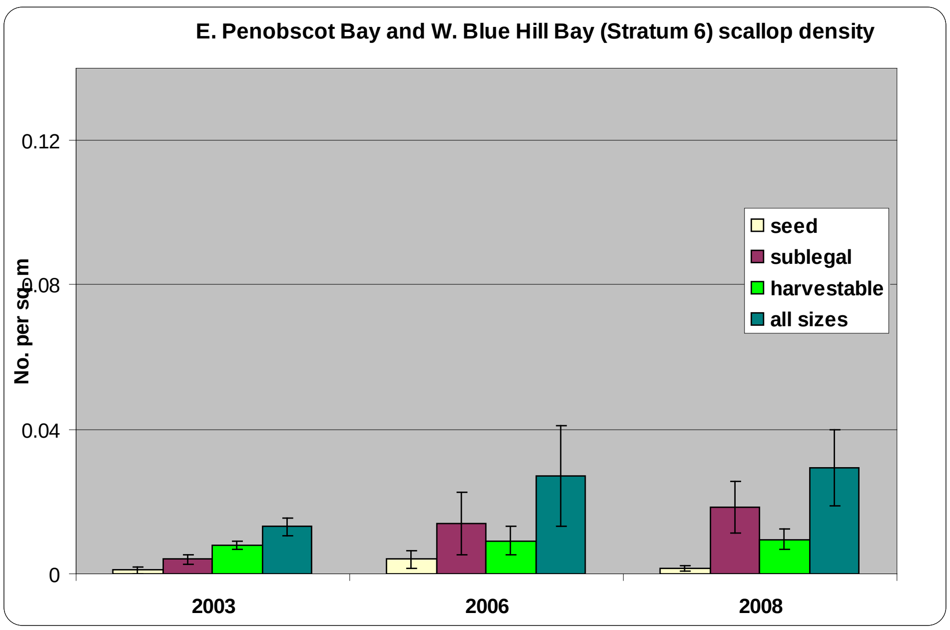
Appendix B5-Figure18. Scallop size frequency (5 mm increments) (top) and mean density (+/- one standard error, unadjusted for dredge efficiency) by size class (bottom), Schoodic Point to Great Wass Island



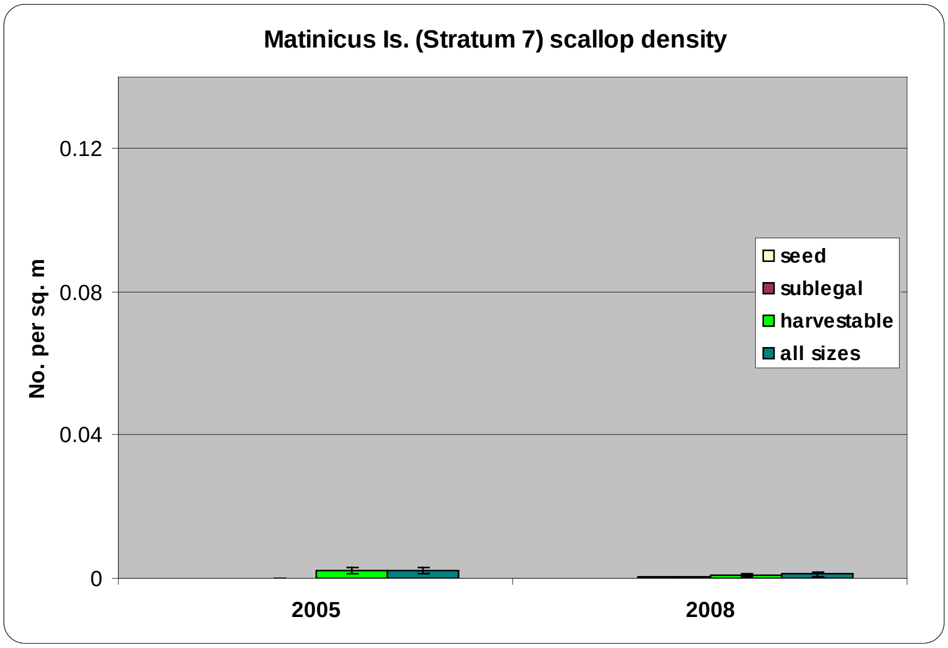


Appendix B5-Figure 19. Scallop size frequency (5 mm increments) (top) and mean density (+/- one standard error, unadjusted for dredge efficiency) by size class (bottom), East Blue Hill Bay and Frenchman Bay.





Appendix B5-Figure 20 . Scallop size frequency (5 mm increments) (top) and mean density (+/- one standard error, unadjusted for dredge efficiency) by size class (bottom), East Penobscot Bay and W. Blue Hill Bay.



Appendix 5-Figure 21. Scallop mean density (+/- one standard error, unadjusted for dredge efficiency) by size class, Matinicus Island

Results from the 2008 survey indicated that scallop abundance has remained low and in some areas slightly declined along the eastern Maine coast (Figures 6-21). These results are similar to reports for adjacent areas of the Canadian coast where landings and survey indices have either declined or remained unchanged since 2006 (Smith et al. 2008). The only region which showed slight improvement was between eastern Penobscot Bay and western Blue Hill Bay (Stratum 6) (Figure 20).

Some small recruitment signals were observed with the presence of seed around Libby Island, Gouldsboro Bay, Union River Bay, South Hancock, Blue Hill Harbor and Southeast Harbor. Three of the locations (Gouldsboro Bay, Blue Hill Harbor and Southeast Harbor) where seed were observed are currently being afforded protection by a series of 3-year area closures implemented by the state prior to the 2009 season. It is hoped the area closures could be particularly beneficial in areas such as these where some resource is present that could be allowed to grow to an optimal size for harvest.

Western Maine: Strata 8-11 (West Penobscot Bay to Kittery)

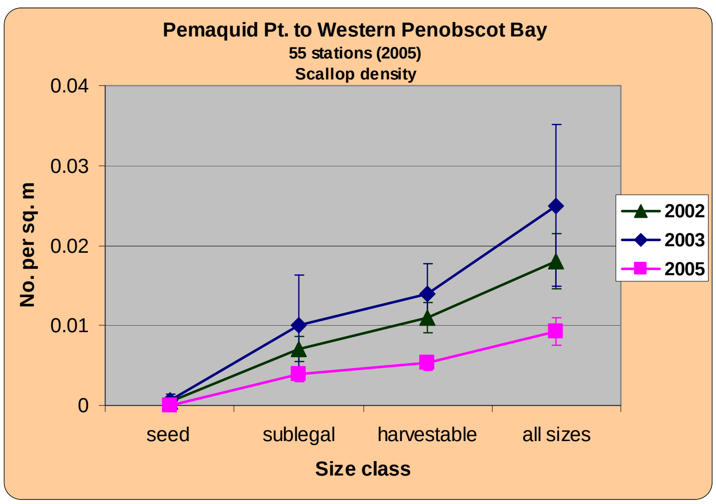
The survey covered these strata in 2005 and 2009. There were 109 tows completed in 2005 and 80 in 2009. The 2005 survey was carried out over 19 vessel days between Nov. 17, 2005 and April 25, 2006. The two contracted vessels were the *F/V North Star* from Portland and the *F/V Sea Ryder* from Spruce Head. The Portland vessel covered strata 10-11 during Nov.-Dec. 2005 and the Spruce Head vessel covered the remaining strata during Feb.-Apr. 2006.

The survey was intended to be performed during late fall, prior to the Dec. 1 opening of the scallop season and after most lobster traps had been removed from the water. For strata 10-11 however, vessel availability and an extended presence of lobster gear in the area precluded completion of the survey before Dec. 1, 2005. In strata 7-9, the survey vessel was not available until January and sampling personnel were not available until February.

Sampling in 2009 was also structured to monitor scallop abundance both inside and outside of the "closed" areas that went into effect in 2009. Tows were distributed to facilitate these areal comparisons. There were also several "fixed" stations sampled which were generally in areas that were considered especially important to monitor on a regular basis. The Piscataqua River area was added to the survey in Stratum 11. Lobster gear was still present in many areas, particularly Casco Bay. Highest 2009 catch rates appear to have been in western Casco Bay and Muscle Ridge Channel and data will be analyzed and a final report on this survey will be completed in 2010.

Results from the 2005 survey indicated that scallop abundance declined across all size categories and throughout all western coastal Maine strata. Overall scallop densities were 49-59% lower than in previous surveys done in 2002 and 2003. The survey zone which comprises Casco Bay had the largest decline.

Casco Bay had the highest density of harvestable scallops (0.006 m^{-2}) observed in the 2005 survey. By comparison the density of harvestable size sea scallops in South Bay (part of Cobscook Bay, the most productive scalloping area in Maine waters) was 0.070 m^{-2} when surveyed in 2006 (Kelly 2007). Highest harvestable density observed in the survey in western Maine was 0.019 m^{-2} in the Small Point to Pemaquid Point stratum in 2003. This survey zone declined to 0.003 m^{-2} in the 2005 survey.

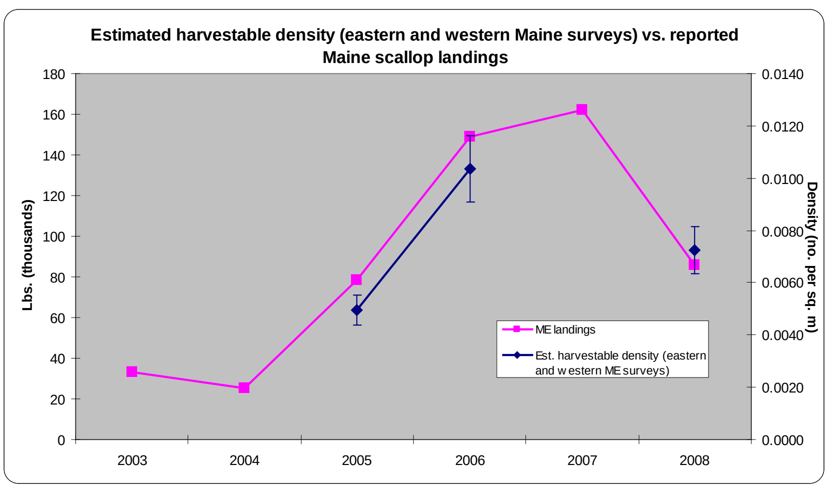


Appendix B5-Figure 22. Mean scallop density by size class, Pemaquid Point to West Penobscot Bay.

Interpretation of these results should be tempered by the fact that the 2005 survey was carried out between Small Point and Matinicus Island well after the commercial scallop season had begun. Although scallop fishing pressure is considered low throughout western Maine (perhaps the Damariscotta River being an exception) it is possible that 2005/2006 season fishing activity could have had an impact on the survey observations. This may account particularly for the size structure of scallops sampled in the Small Point to Pemaquid Point stratum in the 2005 survey. Although sublegal density was similar between 2003 and 2005, harvestable density was much lower in 2005. Fishing removals during 2005/2006 may account for some of the lower density of harvestable scallops observed in the Sheepscot and Damariscotta Rivers.

Eastern/Western Maine survey in relation to landings

As discussed above for Cobscook Bay, Maine scallop landings reports were not required from dealers (and harvesters) until 2008. Reports prior to 2008 were voluntary so landings may not be fully represented. Given those conditions, however, a strong correlation exists when comparing estimated mean harvestable scallop density from the scallop survey in either eastern or western Maine (depending on which area was surveyed in a particular year) and reported Maine landings (Figure 23). This relation is interesting and would not be expected based on the assumption that Maine scallop landings are largely a function of Cobscook Bay. One possible explanation is that the overall condition of the resource is better reflected by abundance within coastal strata rather than from within the rather unique situation of Cobscook Bay. This relation will be of interest to explore following future surveys.



Appendix 5-Figure 23. Mean scallop harvestable density (with standard error, unadjusted for dredge efficiency) estimated by DMR survey in western Maine (2005) and eastern Maine (2006, 2008) in relation to reported Maine landings.

Meat weight modeling

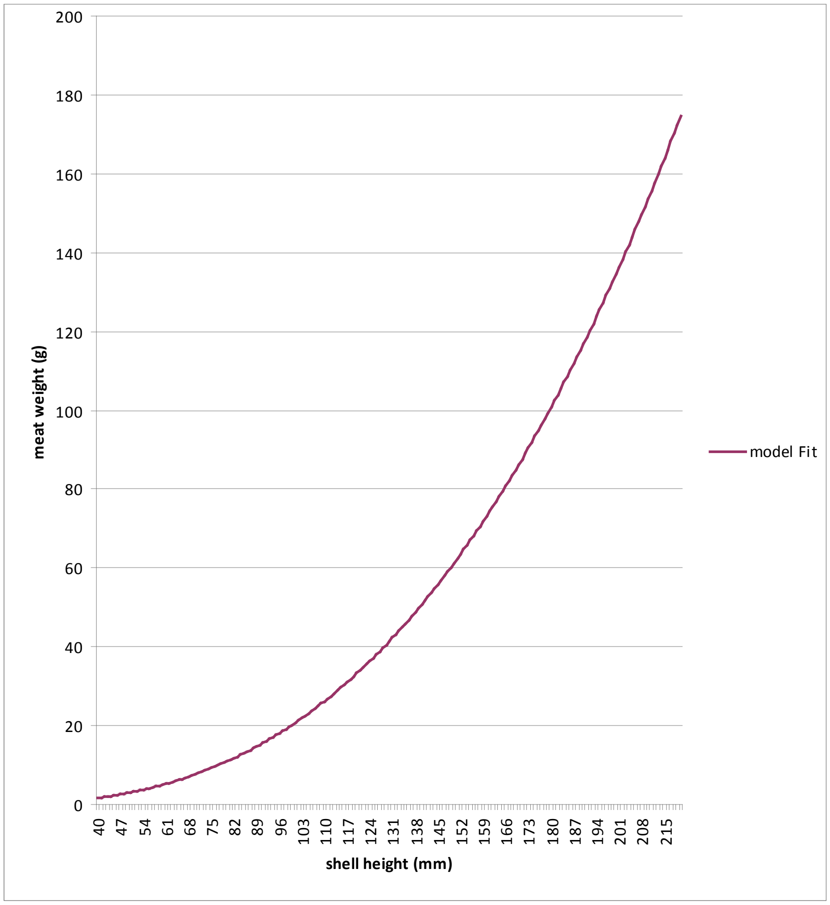
Meat weights were collected from 2,762 scallops during 2005-2008 surveys. Associated with each meat weight were the following parameters: shell height, shell length, shell depth, date, location (station) and depth. Generalized linear mixed models (GLMM) with a log link were used to predict scallop meat weight using the following fixed effects: shell height, shell depth, latitude and depth (Table 2). Random effects were grouped by a variable consisting of the sampling station, or shell height and station. (Modeling courtesy of D. Hennen, Northeast Fisheries Science Center, Woods Hole, MA).

The following model for predicting meat weight had the lowest AIC value:

meat_weight ~ height + depth + lat + s_depth + (height + 1 | station)

Appendix B5-Table 2. Mixed-effect model-building results for prediction of scallop meat weighs in the state waters of Maine, 2005-2008.

Formula	AIC	BIC	logLik	deviance
Maine				
meat_weight ~ height + depth + lat + s_depth + (height + 1 | station)	2083	2136	-1032	2065
meat_weight ~ height + depth + length + lat + (height + 1 | station)	2184	2237	-1083	2166
meat_weight ~ height + depth + s_depth + (height + 1 | station)	2189	2236	-1086	2173
meat_weight ~ height + depth + s_depth + height * depth + (height + 1 | station)	2190	2244	-1086	2172
meat_weight ~ height + s_depth + height * depth + (height + 1 | station)	2190	2244	-1086	2172
meat_weight ~ height + depth + lat + (height + 1 | station)	2239	2286	-1112	2223
meat_weight ~ height + depth + lat + height * depth + (height + 1 | station)	2241	2294	-1111	2223
meat_weight ~ height + depth + length + (height + 1 | station)	2247	2295	-1116	2231
meat_weight ~ height + length + height * depth + (height + 1 | station)	2249	2303	-1116	2231
meat_weight ~ height + depth + length + height * depth + (height + 1 | station)	2249	2303	-1116	2231
meat_weight ~ height + depth + length + lat + (1 | station)	2268	2309	-1127	2254
meat_weight ~ height + lat + (height + 1 | station)	2275	2316	-1130	2261
meat_weight ~ height + length + (height + 1 | station)	2281	2323	-1134	2267
meat_weight ~ height + depth + s_depth + (1 | station)	2298	2333	-1143	2286
meat_weight ~ height + depth + (height + 1 | station)	2305	2346	-1145	2291
meat_weight ~ height + height * depth + (height + 1 | station)	2307	2354	-1145	2291
meat_weight ~ depth + height * depth + (height + 1 | station)	2307	2354	-1145	2291
meat_weight ~ height + depth + length + (1 | station)	2327	2363	-1158	2315
meat_weight ~ height + (height + 1 | station)	2337	2372	-1162	2325
meat_weight ~ height + length + (1 | station)	2363	2392	-1176	2353
meat_weight ~ height + depth + lat + (1 | station)	2407	2443	-1197	2395
meat_weight ~ height + lat + (1 | station)	2443	2473	-1217	2433
meat_weight ~ height + depth + (1 | station)	2471	2500	-1230	2461
meat_weight ~ height + height * depth + (1 | station)	2472	2508	-1230	2460
meat_weight ~ depth + height * depth + (1 | station)	2472	2508	-1230	2460
meat_weight ~ height + (1 | station)	2504	2528	-1248	2496
meat_weight ~ depth + (height + 1 | station)	2729	2764	-1358	2717
meat_weight ~ depth + (1 | station)	11467	11491	-5730	11459



Appendix B5-Figure 24. Scallop shell height vs. meat weight relationship based on Maine (2005-2008) data at 22 m (12 fathoms) in depth and 44°N latitude.

Conclusions

Results from the surveys of ME state eaters indicate that scallop abundance has remained low and in some areas has slightly declined along the eastern Maine coast. Some recruitment signals were observed, however, in the most recent eastern Maine survey (2008), particularly in the zone between eastern Penobscot Bay and western Blue Hill Bay. Cobscook Bay, at the far eastern end of the Maine coast, remains the most heavily fished and productive area in Maine waters. The 2007 estimate of harvestable biomass 128 mt (281.3 thousand lbs) of meats in Cobscook Bay was 99.4% higher than the previous year. Overall western Maine scallop densities were 49-59% lower in 2005 than in previous surveys done in 2002 and 2003. The survey zone which comprises Casco Bay had the largest decline in 2005.

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Appendix B6: An assessment of the sea scallop resource in the Northern Gulf of Maine management area.

Samuel B. Truesdell (University of Maine, Orono), Kevin H. Kelly (Maine Department of Marine Resources, W. Boothbay Harbor, ME), Catherine E. O'Keefe (University of Massachusetts, Dartmouth), and Yong Chen (University of Maine, Orono).

The sea scallop fishery in the Northern Gulf of Maine (NGOM) occurs in federal waters and is managed by the New England Fishery Management Council. The NGOM resource and associated fishery are locally important but amount to a small portion of the total stock and landings. The fishery is managed by TAC independently of the rest of the EEZ sea scallop stock. In particular, management of the NGOM fishery does not involve biological reference points as targets or thresholds. A cooperative survey was carried out by the Maine Department of Marine Resources and the University of Maine in June-July, 2009. The best estimate based on survey results indicates that the biomass of NGOM sea scallops targeted by the fishery (102+ mm or 4+ in shell height) was approximately 103 mt of meats during 2009 with a 95% confidence interval ranging from about 53 to 186 mt. Landings during 2009 amounted to approximately 7 mt. The best estimate of exploitation rate (reported landings in weight / estimated biomass) in the NGOM during 2009 was 0.065, with a 95% confidence interval ranging from 0.035 to 0.12. These estimates are based on density estimates from the survey assuming a range of survey dredge capture efficiency of 40%. NGOM biomass was relatively low during 2009, although small (10-50 mm) "seed" scallops were abundant at two stations on Platts Bank.

Background

Sea scallops (*Placopecten magellanicus)* have been an important resource in the Gulf of Maine coastal region since before European settlement. Initially supplementing the diets of early European settlers and Native Americans (Bourne 1964), a commercial scallop fishery eventually developed in the 1880s (Dow 1956, Bourne 1964, Baird 1967). The Gulf of Maine fishery expanded after World War I (Dow 1971), although fishing effort remained mainly inshore until 1950, when some fishing began in more offshore areas (Dow 1956). Since then, the scallop fishery in the Gulf of Maine has undergone substantial fluctuations with landings ranging from hundreds of thousands to millions of pounds within as little as a three year period (Figure 1).

The recent Amendment 11 to the New England Fishery Management Council Sea Scallop Fishery Management Plan (New England Fishery Management Council 2008) created a separate limited entry program for general category fishing in the Northern Gulf of Maine (NGOM) management area (Figure 2). The program includes a yearly NGOM total allowable catch (TAC; currently 70,000 lbs.) and a daily possession limit of 200 lbs. (New England Fishery Management Council 2008). The effective date of the new management regime was June 1, 2008.

The 2008 NGOM TAC was set based on 2000-2006 landings from federal waters of the Gulf of Maine (New England Fishery Management Council 2008) because information on stock abundance in this area was minimal. In June-July 2009, the Maine Department of Marine Resources (DMR) and the University of Maine (UM) collaborated under the FY 2008 Scallop Research Set-Aside Program to survey this new management area, with the goal of estimating the harvestable scallop biomass and providing information that might be used in updating the

TAC. The survey was carried out aboard the F/V *Foxy Lady II* out of Stonington, ME under contract with the DMR.

Methods

The NGOM was divided into five areas for the purposes of this survey, referred to here (from east to west) as Machias Seal Island (Area 1), Mt. Desert Rock (Area 2), Platts Bank (Area 3), northern Stellwagen Bank (Area 4), and Cape Ann (Area 5; Figure 2). Selection of these areas was based on previous offshore Gulf of Maine scallop surveys (Spencer 1974, Serchuk and Rak 1983, Serchuk 1984, Serchuk and Wigley 1984); recent (2000-2008) vessel trip reports (VTR) indicating the location and magnitude of scallop catches by vessels fishing in federal portions of the Gulf of Maine; recent Maine/New Hampshire inshore trawl survey data (S. Sherman, DMR, pers. comm.); and input from two Maine-based federally-permitted scallop fishermen with experience fishing these areas. VTR data, in particular, indicate that most scallop catches by federally-permitted vessels during 2000-2008 were from Areas 4 and 5.

The survey followed an adaptive two-stage random stratified design (Francis 1984) in areas 4 and 5. These regions were delineated into high, medium, and low density sub-areas based on expected survey catch in order to increase sampling precision. The stratification was based on 2000-2008 VTR data and input from the survey captain and an experienced federally-permitted scallop fisherman. Forty tows were allocated to the first stage among the three sub-areas. After the first survey stage, the within sub-area variance was calculated. Using this variance in combination with the area size, the number of tows allocated to each sub-area in stage 2 was calculated according to the method used by Francis (1984).

Area 2 was stratified into high and low densities. However, because of its large size, the survey in this area was only a single stage. Areas 1 and 3 were not divided into subareas due to low expected scallop densities.

One hundred and ninety-six stations were occupied in total. Tows lasted either five or seven minutes depending on the bottom type and amount of fixed fishing gear in the area. The survey dredge was a 7 ft New Bedford style drag with 2 in rings, 1.75 in head bale, 3.5 in twine top, 10 in pressure plate and rock chains. The dredge had no liner.

At each tow location, all species were identified and counted. Excluding tows on Platts Bank where large numbers of scallop seed were caught, survey catches were low enough that approximately 98% of all scallops were measured for shell height (SH) and about 50% of measured scallops were also sampled for their meat weight (MW) for use in developing a SH to MW relationship.

Results

The most evident features of the NGOM survey length frequency distribution (Figure 3) are the dominance of scallops under 50 mm on Platts Bank and the size class distribution differences between the eastern and western NGOM.

Large numbers of scallop seed were found on Platts Bank, most of which were caught at two stations on the eastern side of the bank (estimated at over 15,000 individuals between the two tows). Some seed scallops were found in other areas but at substantially lower densities.

Another important finding regarding the length frequency distribution is the difference in breadth of size distribution between the eastern and western NGOM. The Cape Ann and Stellwagen Bank survey areas showed a broader size class distribution (approximately 50 – 150 mm) than those in the eastern NGOM (Platts Bank, Mt. Desert Rock and Machias Seal Is.;

Figure 3). This indicates that the western NGOM has had, in general, consistent recruitment and that scallops are able to settle and survive during most years. In contrast, the eastern NGOM tends toward episodic recruitment when conditions are favorable and the populations at these sites are composed primarily of a single size class. See Figure 4 for by-tow length frequency distribution.

Meat weights

The estimated meat weights used to determine the NGOM biomass estimates were based on area-specific shell height-meat weight (SHMW) relationships for the eastern and western NGOM. Meat weight was modeled as a function of shell height assuming multiplicative error structure as:

$$MW_i = \alpha SH_i^{\beta} e^{\varepsilon_i} .$$

SHMT relationships varied considerably over the NGOM survey area (Figure 5). The largest meats were found on northern Stellwagen Bank, followed by Cape Ann and Mt. Desert Rock. The lowest meat weights were found on Platts Bank; however, this was based on a sample size of only 8 scallops. Low meat weights from some eastern Maine areas have been noted in previous reports (Serchuk and Rak 1983, Schick and Feindel 2005).

Biomass estimates

Bootstrapped biomass mean and 95% confidence interval estimates were calculated (1,000 replications) using the "NMFSsurvey" package version 1.0-2 written by Stephen Smith (Canada DFO) in R version 2.8.1. This package allows for various combinations of bootstrap mean and 95% confidence interval calculations. The available bootstrap mean methods are: naïve, rescaling and bootstrap-with-replacement (BWR) and the available confidence interval methods are: percentile (PCT), bias-corrected (BC), and bias-corrected-and-adjusted (BCa).

The bootstrap functions were run under each combination of bootstrap mean and 95% confidence interval calculations at assumed dredge capture efficiency estimates of 30%, 40%, and 50% (Figures 6 and 7). The middle estimate of 40% efficiency was selected as the best estimate because it is close to an estimate by the DMR of 43.6% measured in Cobscook Bay, Maine in 2006 (Kelly 2007). Figures 6-7 show that harvestable biomass was estimated at around 100 mt with absolute maximum confidence intervals from 39.7 (50% efficiency and BWR/PCT bootstrap approach) to 320 mt (30% efficiency and naïve/BCa bootstrap approach). Harvestable biomass was calculated assuming scallops under 4 in SH are too small for commercial boats to regularly target, so only scallops larger than 4 in SH were included in the estimates. The bootstrap means were stable for all efficiencies and all bootstrap methods, though there is some variation in confidence intervals among bootstrap approaches, especially at the upper bounds.

For ease of explanation, and because similar results were found under each combination of methods, the BWR/BC combination is used in the subsequent sections. This combination was found by Smith (1997) to be acceptable for estimating haddock numbers and 95% confidence intervals in a stratified random survey.

Regional biomass estimates

Figures 8 and 9 indicate that Area 1 has the highest mean biomass, though Area 3 has the largest upper confidence level bound (greater than 200,000 kg at 30% dredge efficiency) due to low sample size and high sample variability. Density calculations also show that scallops in Area 1 appear more abundant per unit area than in any of the other strata (although a substratum in area 4 had the highest overall density). It is therefore surprising that federal vessel trip reports indicate low fishing effort in this region. Possible explanations include the high density of fixed gear in the region and poor meat quality. This area is an important lobster fishing ground and there are large numbers of lobster traps present. During the NGOM survey, alternate stations had to be used and tow durations had to be shortened in this region so that fixed gear was not damaged. Due to poor meat quality (Figure 5), more shucking effort is required to obtain the same amount of meat as in the more productive western NGOM.

Area 3 has the second highest bootstrapped mean biomass at 40% dredge efficiency (Figure 8), but because of limited time for sampling (16 tows) and high degree of variability in catch, the 95% confidence interval ranges from close to zero to over 150,000kg. This variability, along with the large year class of seed scallops, makes Platts Bank a high priority for subsequent NGOM surveys.

The Mt. Desert Rock area (Area 2) had few scallops. Historically there has been some fishing in this region and the Maine fishery has its origins in Mt. Desert Island inshore waters (Smith 1891), but little activity has been recorded in Area 2 in recent years.

The two western NGOM areas (4 and 5) exhibit relatively low biomass (Figure 8) but support most of the fishing activity. The limited fixed gear and good meat condition (Figure 5) are probably the two main contributors to the higher rate of fishing. The high sampling rate (60 tows in each of the two regions) increased precision over the other areas.

Exploitation rates

The 2009 estimated exploitation rate for the NGOM at 40% dredge efficiency was 0.065, with a 95% confidence interval ranging from 0.035 to 0.12 (based on the BWR/BC method; Figure 10). Landings are based on dealer and vessel reports and were retrieved from the NMFS Northeast Regional Office website.[4]

The exploitation estimates were somewhat sensitive to the assumed capture efficiency level. The mean exploitation rate for assumed efficiency of 30% is 0.049 and the mean for assumed efficiency of 50% is 0.080. The range in estimated confidence intervals (the lower bound of the 95% confidence interval at 30% efficiency and the upper bound of the 95% confidence interval at 50% efficiency) was from 0.027 to 0.15 (Figure 10).

The exploitation rate may be higher in some regions, particularly in Areas 4 and 5 in the western NGOM. However, these rates were not able to be estimated due to data confidentiality (VTR reports were for less than 3 vessels).

Platts Bank

The Platts Bank survey area (Area 3; Figure 11) deserves special consideration because two sample locations saw numbers of seed scallops in the thousands (see Figure 4 tows SM3C04 and SM3C10). These densities were much larger than elsewhere in state or federal waters of the Gulf of Maine. The DMR/UM survey had relatively few (16) tows in Platts Bank because. Although productive in the past, Platts Bank has seen little fishing in recent years so high densities were not anticipated.

The University of Massachusetts School for Marine Science and Technology (SMAST) also surveyed Platts Bank in 2009 (Figure 12). The SMAST survey used a drop pyramid with two different cameras which photographed the bottom at each sample location (see Stokesbury and Harris 2006 for details). Scallop densities and other individual and population statistics were

[4] http://www.nero.noaa.gov/ro/fso/Reports/ScallopProgram/NGOMReport%2020100223.pdf

estimated from the photos. The DMR/UM survey occurred on July 28th and the SMAST survey on August 12 and 13, 2009. The two surveys complemented each other because the DMR/UM survey was able to cover a large area per station and the SMAST survey was able to sample a large number of stations distributed across the area.

As the survey areas were delineated differently between the two projects, biomass estimates are difficult to compare. Therefore, only densities and length frequency data are used in comparing results. Mean scallop densities from the two surveys were almost identical: SMAST estimated 1.87/m^2 and DMR/UM estimated 1.81/m^2 (table 1). The confidence intervals, however, were quite different. The SMAST confidence interval is symmetric and estimated assuming a normal distribution while the DMR/UM mean (assuming 40% dredge efficiency) was bootstrapped as described above. Despite the differences in computation of confidence intervals, the main reason the SMAST confidence interval is smaller is that the sampling design allowed for many more sampling locations. The two surveys generally agreed on the spatial distribution of scallop density (Figures 11 and 12) with highest densities on the eastern side of Platts Bank.

High scallop densities on Platts Bank were the result of a recruitment event. It is not known, however, whether this will result in increased fishing activity in the future. The scallops of harvestable size that were sampled on the DMR/UM survey had very low SHMW relationships but only 8 scallops larger than 4 inches were sampled (see Figure 5). Two reasons potentially explain this poor meat quality. One explanation is that Platts Bank is currently poor habitat for scallops. The other explanation is that the meats sampled were simply from older, poorer-condition scallops and that the new recruitment class will potentially have better meats.

The DMR/UM and SMAST shell height composition data are compared in Figures 13 and 14. Compared to the SH measurements from the SMAST large camera, the DMR/UM distribution is shifted somewhat to the left. However, compared to the SMAST digital still camera, the DMR/UM distribution is shifted only slightly to the left. This may be due to the timing of the surveys. The DMR/UM survey took place in late July 2009 and the SMAST survey in mid-August 2009, so the difference between the DMR/UM and SMAST digital still camera SH frequencies could be attributed to growth over the period between the surveys.

When the densities, length frequencies, and spatial distributions are considered, the two surveys compare well. It appears that the DMR/UM survey achieved a large enough sample size to well-characterize the Platts Bank population. Ideally, however, more tows will be included in the future to increase precision. In addition, the SMAST survey was able to estimate the length frequency distribution observed by the DMR/UM survey with their digital still camera without bringing animals to the surface, assuming the slight shift in the SMAST distribution is due to growth.

Recruitment dynamics are unclear in the NGOM. An interesting note is that little recent recruitment was observed in the southwestern NGOM (Cape Ann and Stellwagen Bank). It is possible that oceanographic conditions contributing to recruitment on Platts Bank also reduced larval input to southwestern NGOM.

Conclusions

The 2009 DMR/UM survey confirmed what recent landings data suggest: scallop biomass is currently low in the NGOM management area. NGOM scallops are not heavily fished as the exploitation rate (catch/biomass) is estimated at approximately 0.07. The survey found significant biomass in the Machias Seal Is. area (close to 50,000 kg), an area that is hardly fished probably due to the high concentration of fixed gear and poor meat quality. This area contributes greatly to the low exploitation rate because of its size and lack of fishing. The western Gulf of Maine (Cape Ann and Stellwagen Bank areas) probably have higher exploitation rates. However, rates for these areas could not be estimated due to confidentiality constraints (VTR reports were for fewer than 3 vessels).

The high densities of scallop seed noted on Platts Bank by both the DMR/UM and SMAST surveys could prove important once those scallops recruit to the fishery. The poor meats encountered on Platts Bank by the DMR/UM survey also leave open the possibility that while densities on Platts Bank may be very high, meat quality may be low. Few samples were taken on Platts Bank, however, so the poor meats are not necessarily representative.

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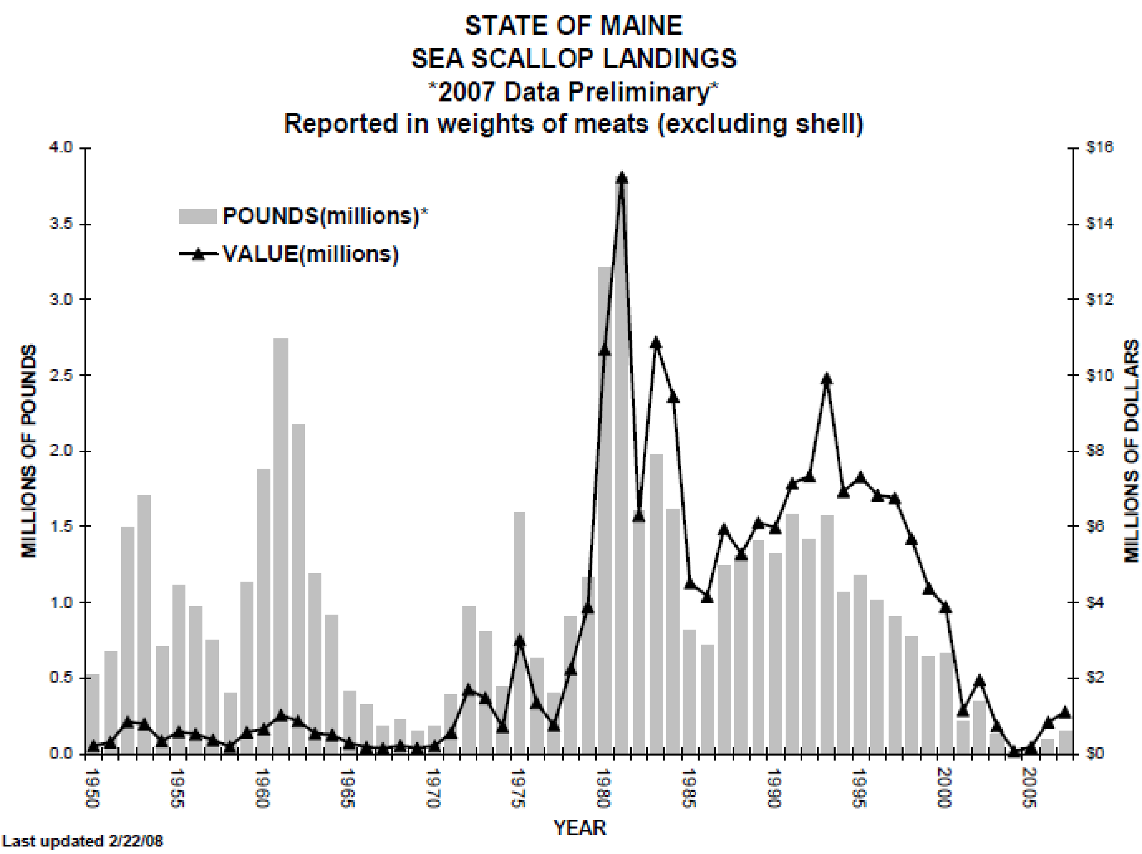
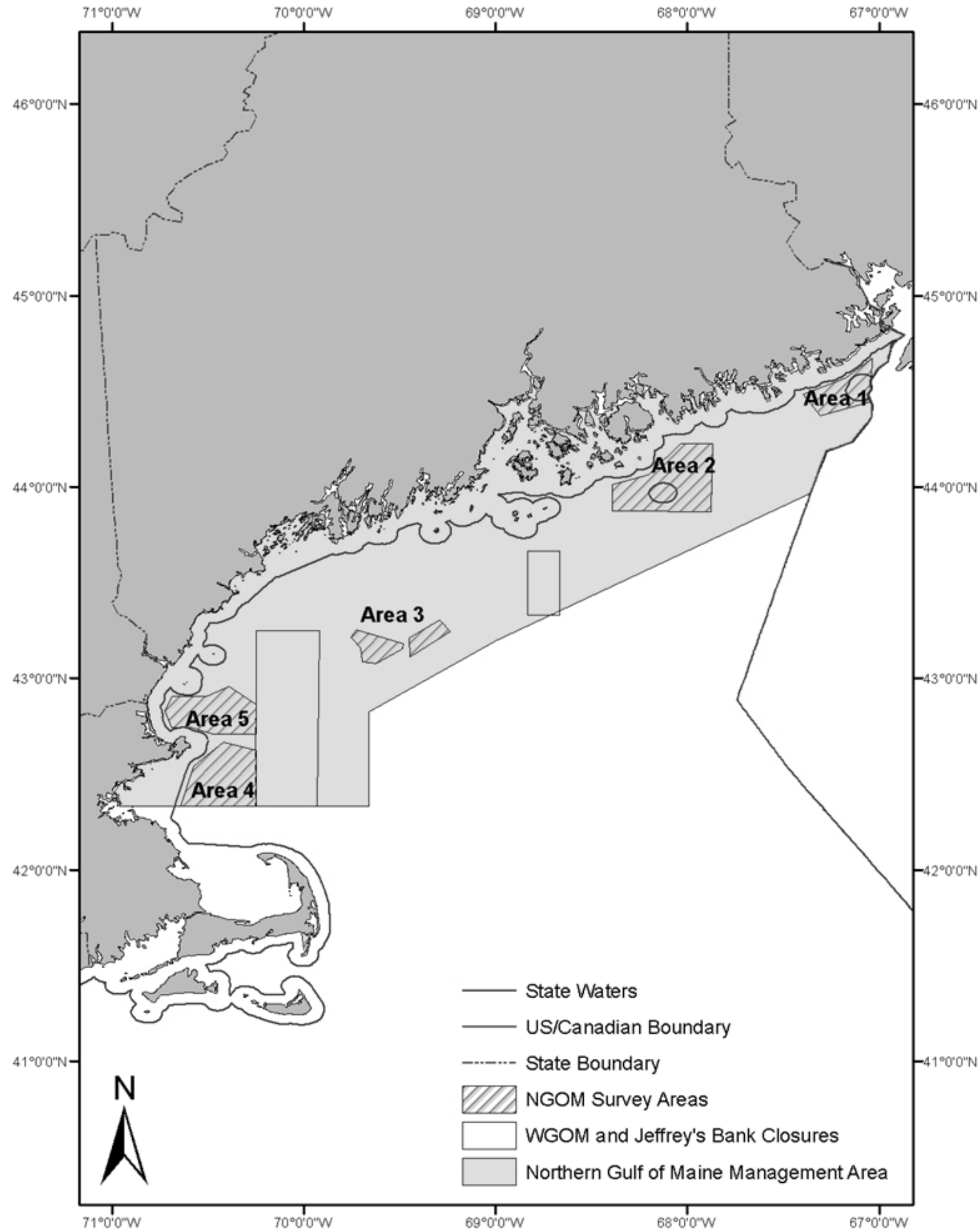
Appendix 6-Table 1. Estimated scallop density (all size classes) on Platts bank for the DMR/UM and SMAST surveys in 2009.

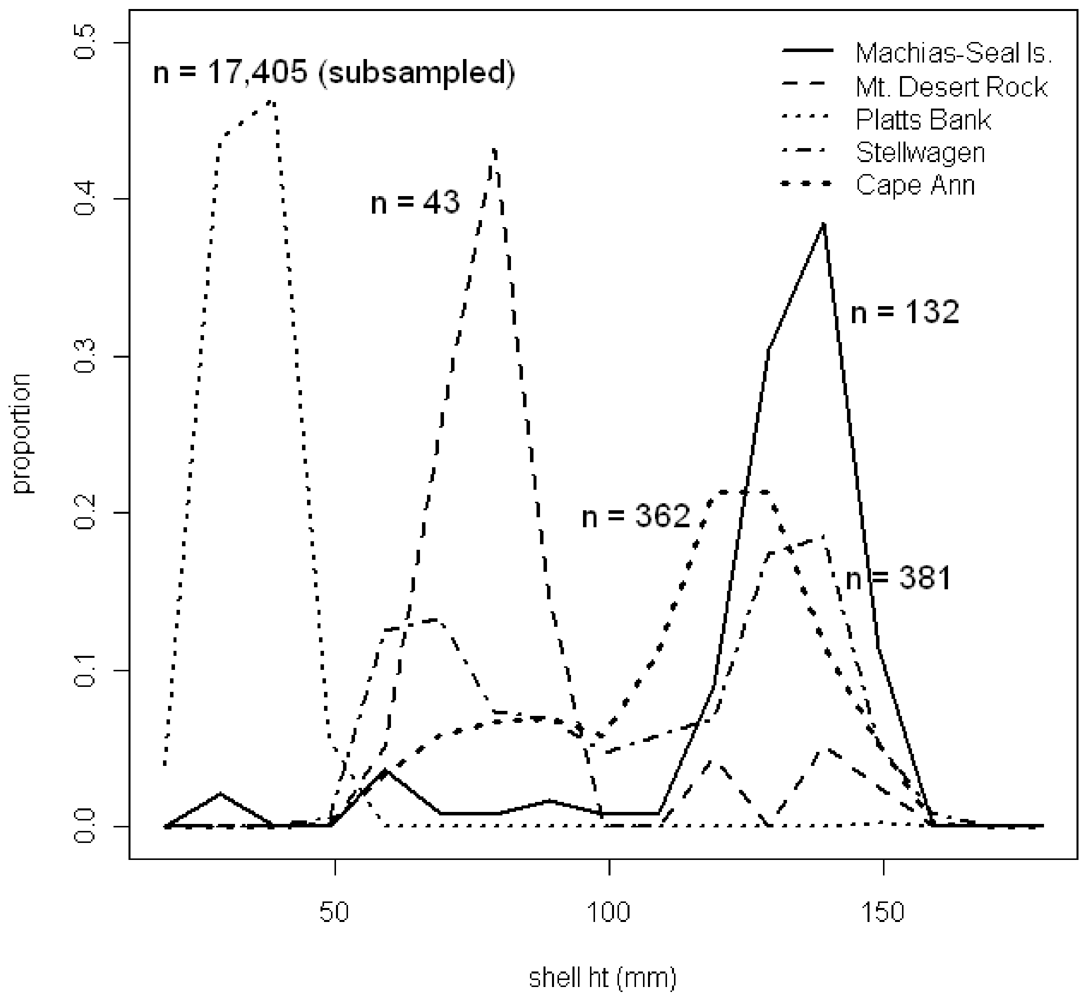
Survey	Mean Density	95% confidence interval
SMAST	1.87	(0.674 , 3.066)
DMR/UM	1.805	(0.014 , 5.071)



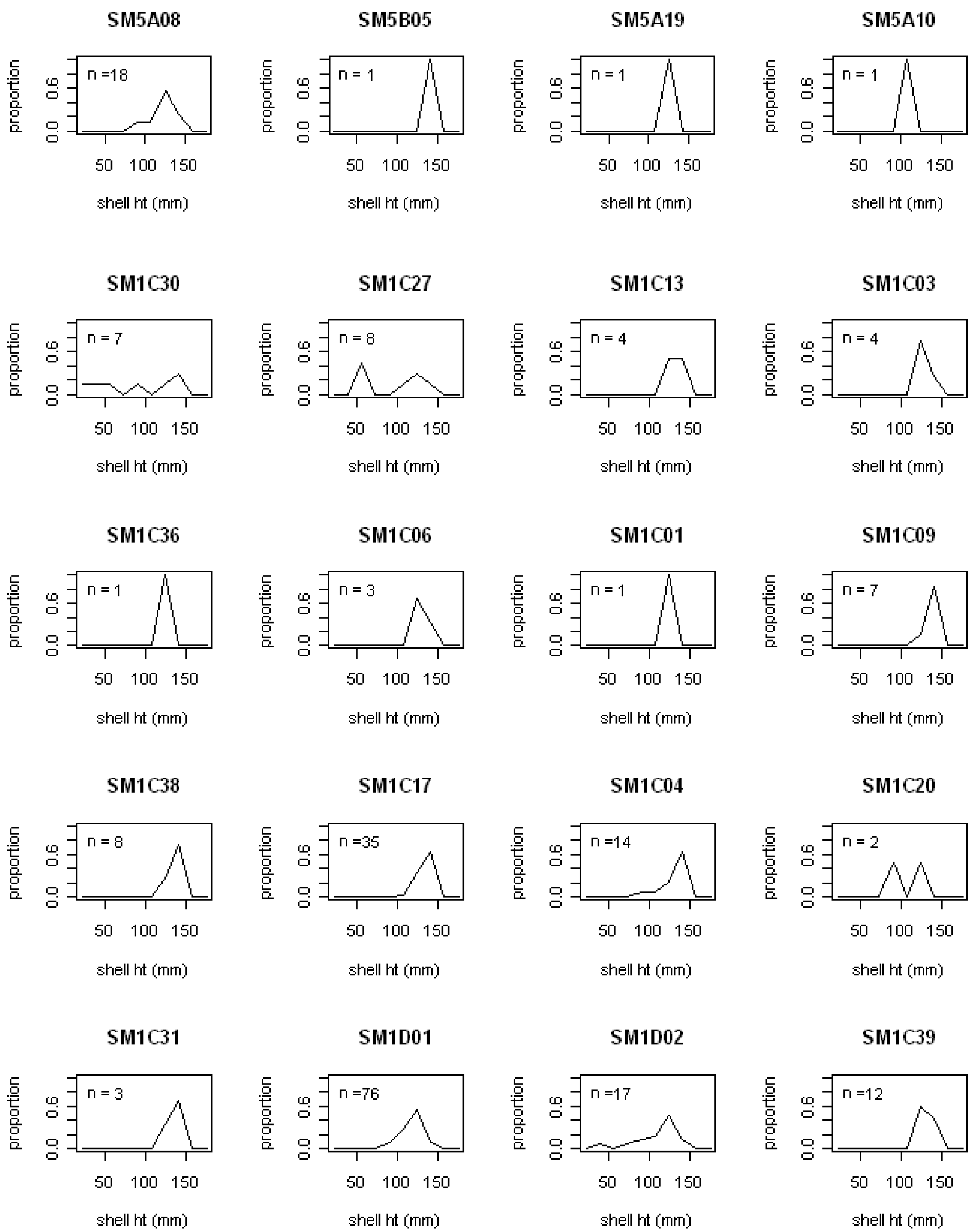
Appendix B6-Figure 1. Maine scallop landings (inshore and offshore) and ex-vessel revenues 1950 through 2007.

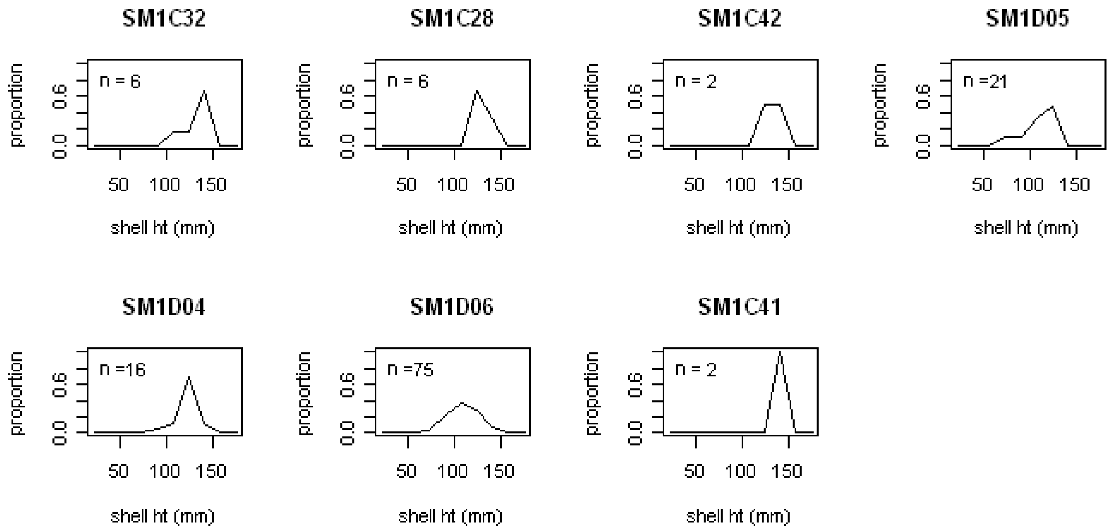


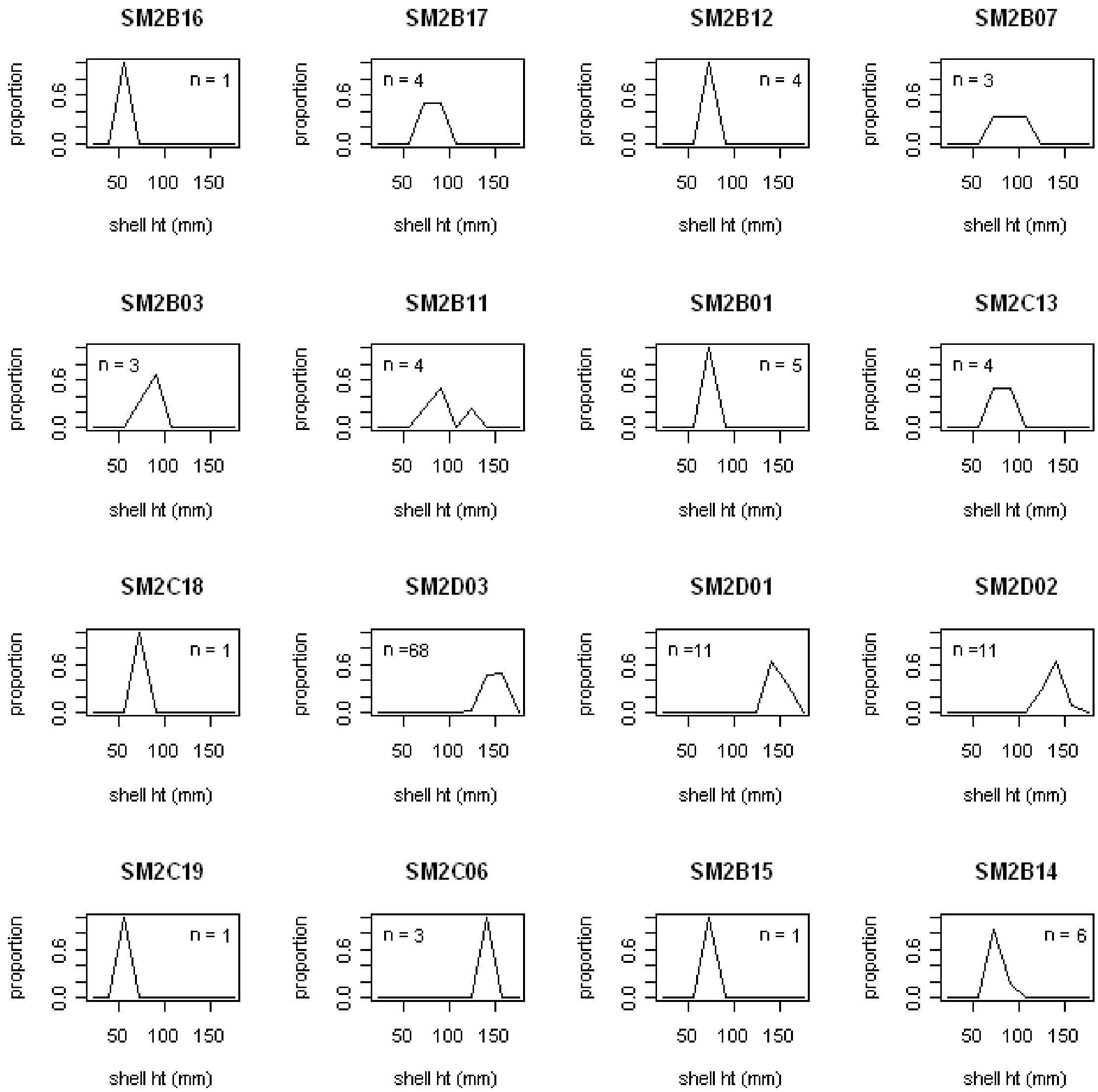
Appendix B6-Figure 2. The NGOM management area was divided into 5 regions for the DMR/UM 2009 survey. In numerical order the areas are: Machias Seal Island, Mt. Desert Rock, Platts Bank, Stellwagen Bank and Cape Ann.

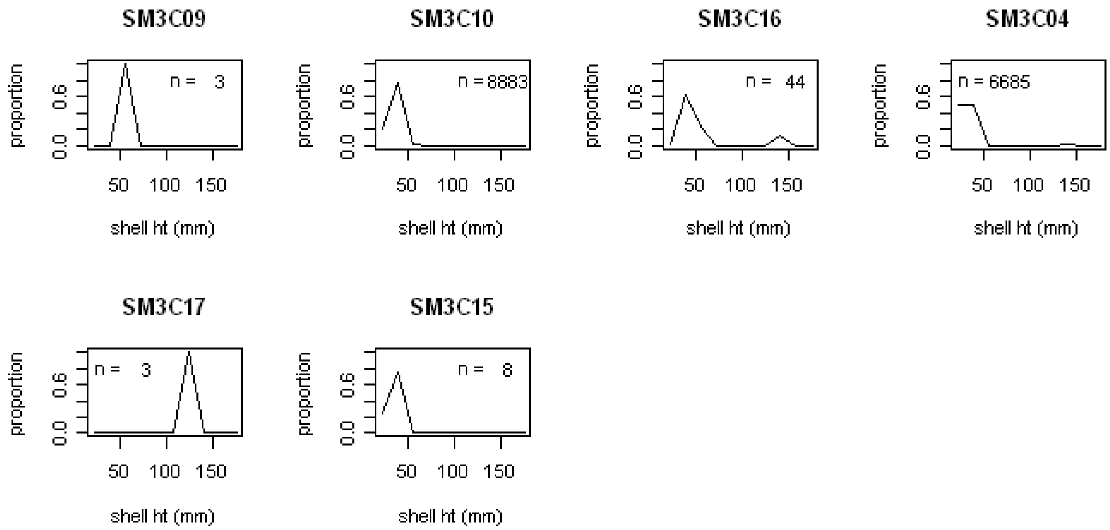


Appendix B6-Figure 3. The NGOM length frequency distribution estimated by the DMR/UM survey. The western Gulf of Maine (Stellwagen Bank and Cape Ann) has a much broader size class distribution. Large numbers of seed scallops were found on Platts Bank.


SM5A08
n =18
SM5B05
n = 1
SM5A19
n = 1
SM5A10
n = 1
SM1C30
n = 7
SM1C27
n = 8
SM1C13
n = 4
SM1C03
n = 4
SM1C36
n = 1
SM1C06
n = 3
SM1C01
n = 1
SM1C09
n = 7
SM1C38
n = 8
SM1C17
n =35
SM1C04
n =14
SM1C20
n = 2
SM1C31
n = 3
SM1D01
n =76
SM1D02
n =17
SM1C39
n =12
proportion
0.0
0.6
50
100
150
shell ht (mm)


SM1C32
n = 6
SM1C28
n = 6
SM1C42
n = 2
SM1D05
n =21
SM1D04
n =16
SM1D06
n =75
SM1C41
n = 2
proportion
0.0
0.6
50
100
150
shell ht (mm)


SM2B16
n = 1
SM2B17
n = 4
SM2B12
n = 4
SM2B07
n = 3
SM2B03
n = 3
SM2B11
n = 4
SM2B01
n = 5
SM2C13
n = 4
SM2C18
n = 1
SM2D03
n =68
SM2D01
n =11
SM2D02
n =11
SM2C19
n = 1
SM2C06
n = 3
SM2B15
n = 1
SM2B14
n = 6
proportion
0.0
0.6
50
100
150
shell ht (mm)


SM3C09
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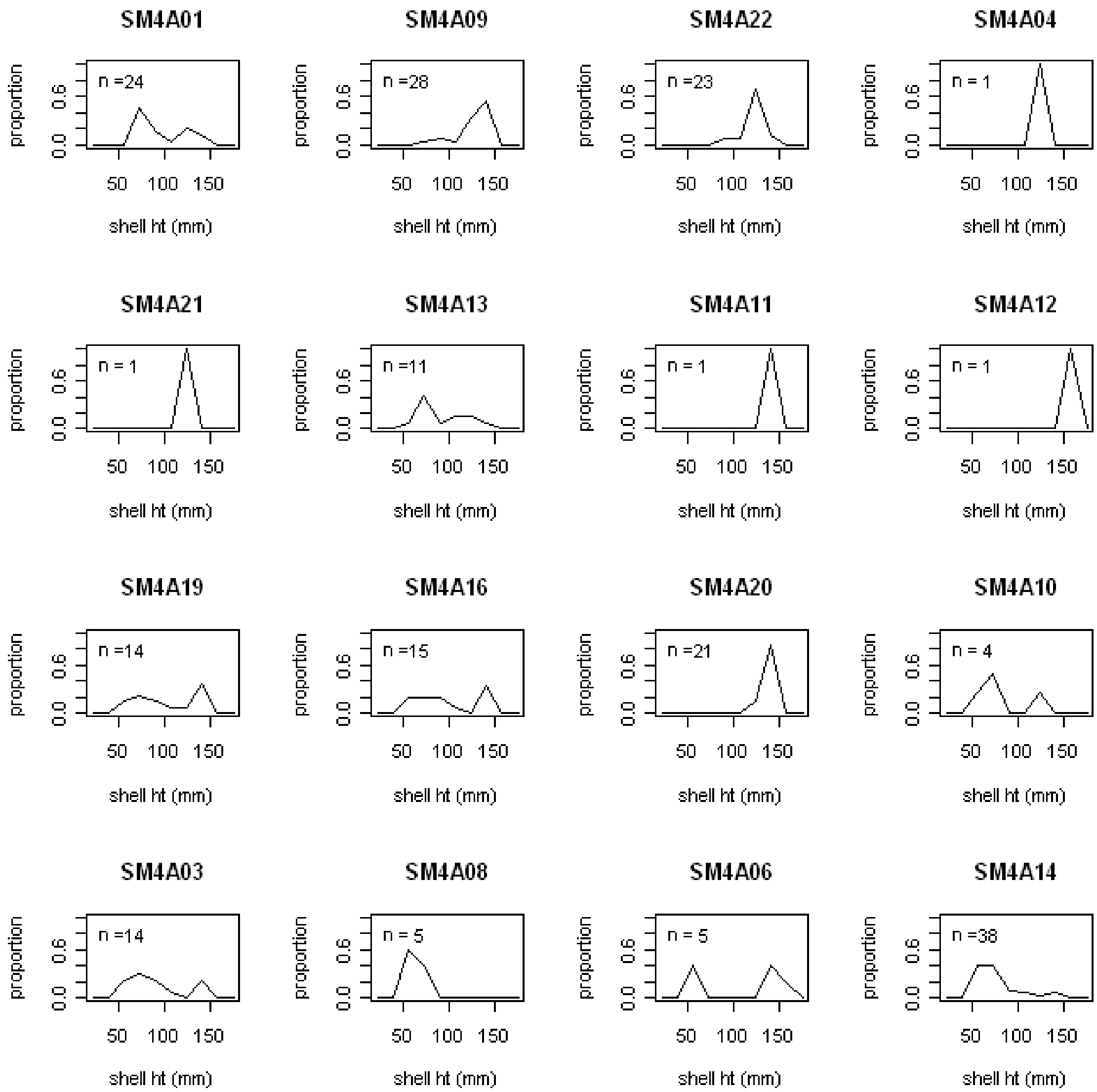
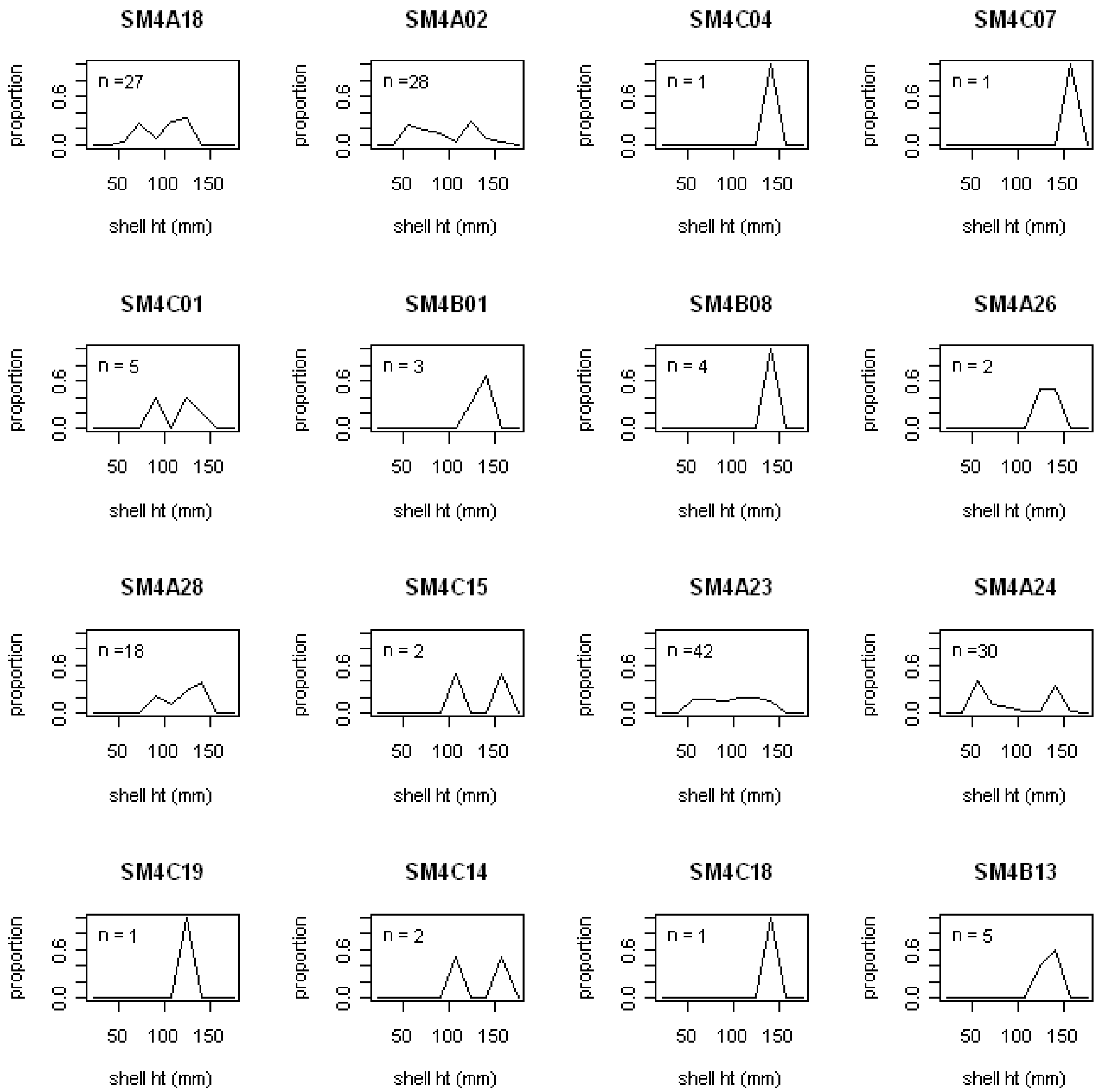
SM3C10
n = 8883
SM3C16
n = 44
SM3C04
n = 6685
SM3C17
n = 3
SM3C15
n = 8
proportion
0.0
0.6
50
100
150
shell ht (mm)


SM4A01
n =24
SM4A09
n =28
SM4A22
n =23
SM4A04
n = 1
SM4A21
n = 1
SM4A13
n =11
SM4A11
n = 1
SM4A12
n = 1
SM4A19
n =14
SM4A16
n =15
SM4A20
n =21
SM4A10
n = 4
SM4A03
n =14
SM4A08
n = 5
SM4A06
n = 5
SM4A14
n =38
proportion
0.0
0.6
50
100
150
shell ht (mm)


SM4A18
n =27
SM4A02
n =28
SM4C04
n = 1
SM4C07
n = 1
SM4C01
n = 5
SM4B01
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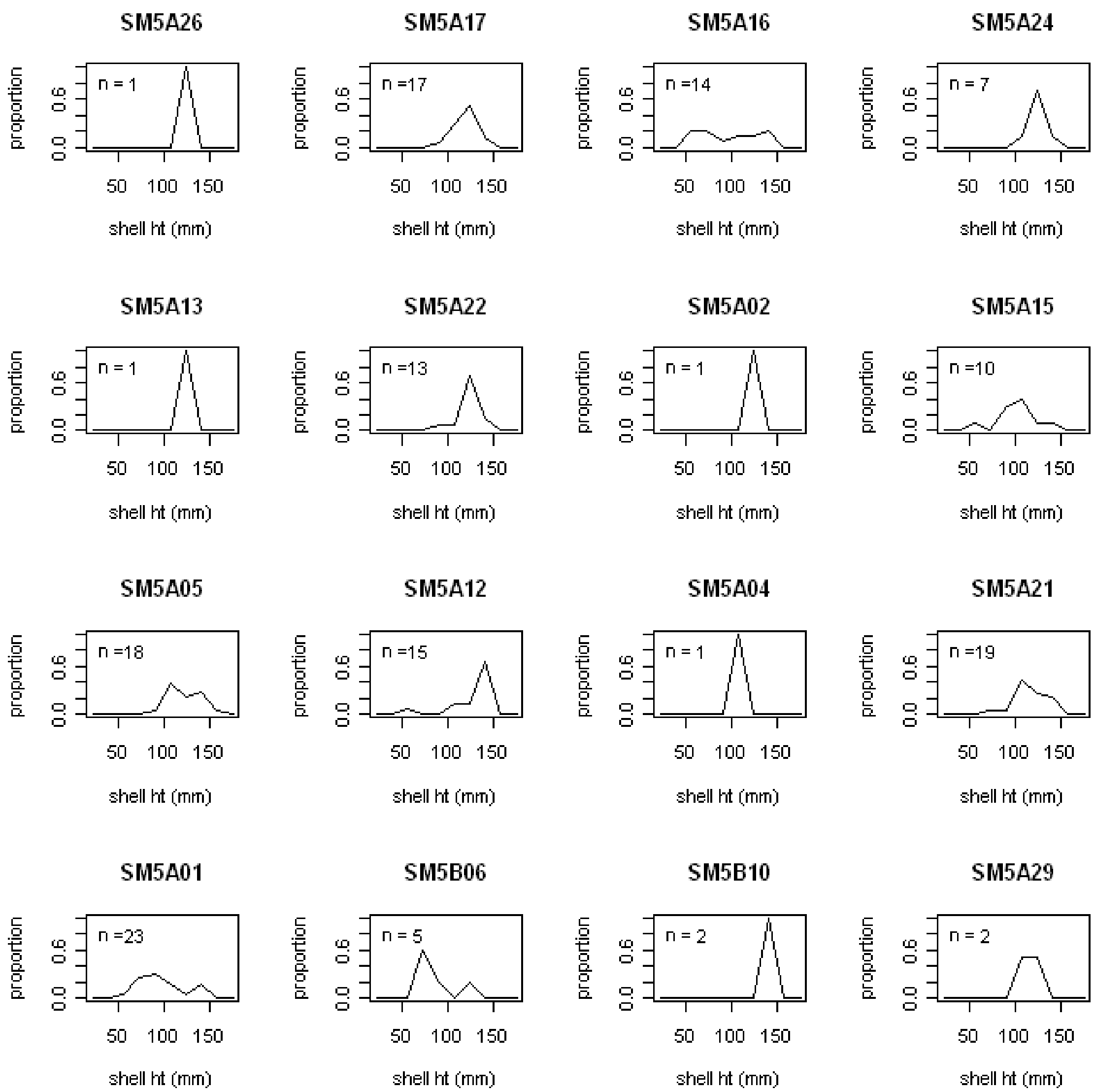
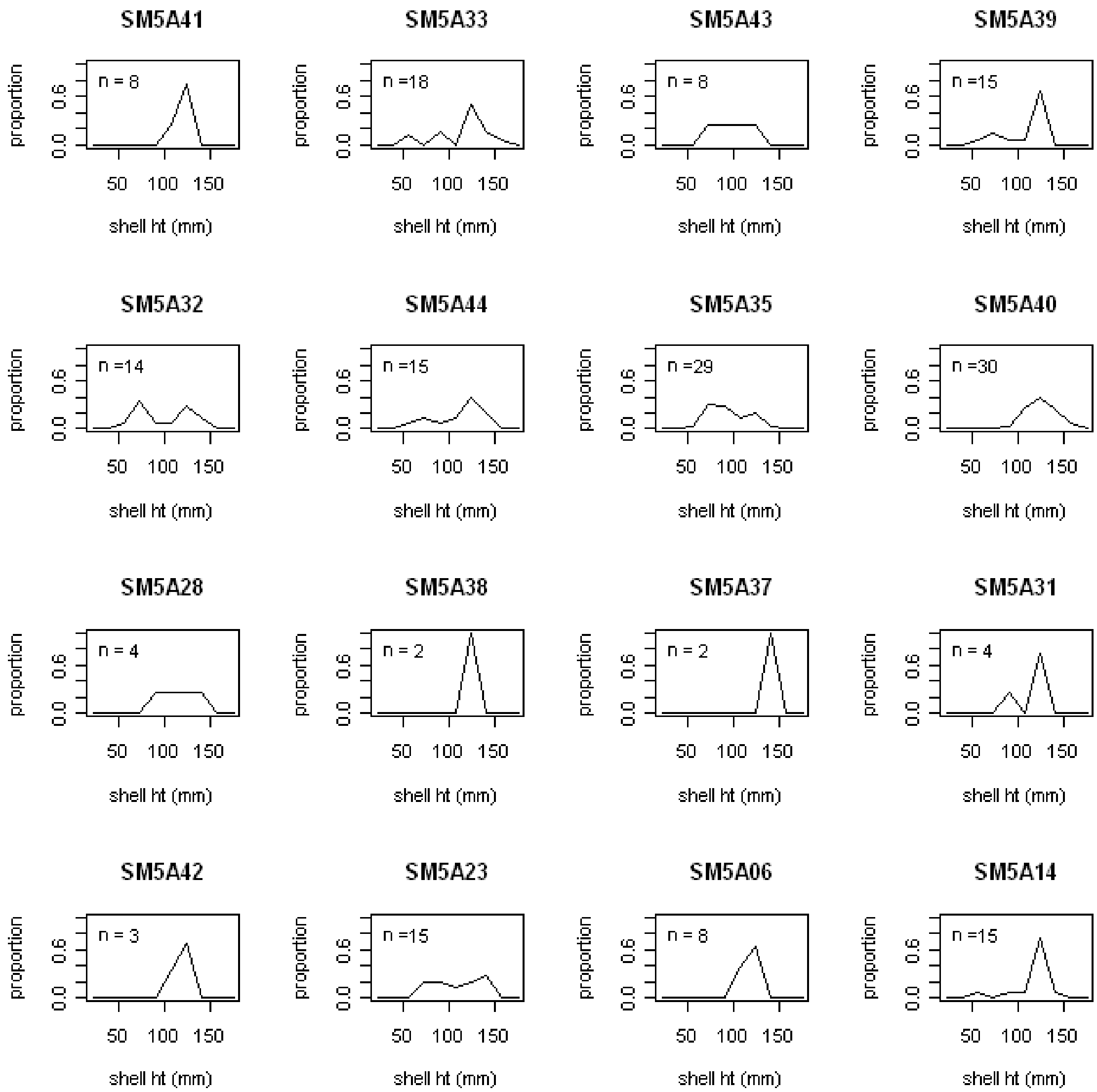
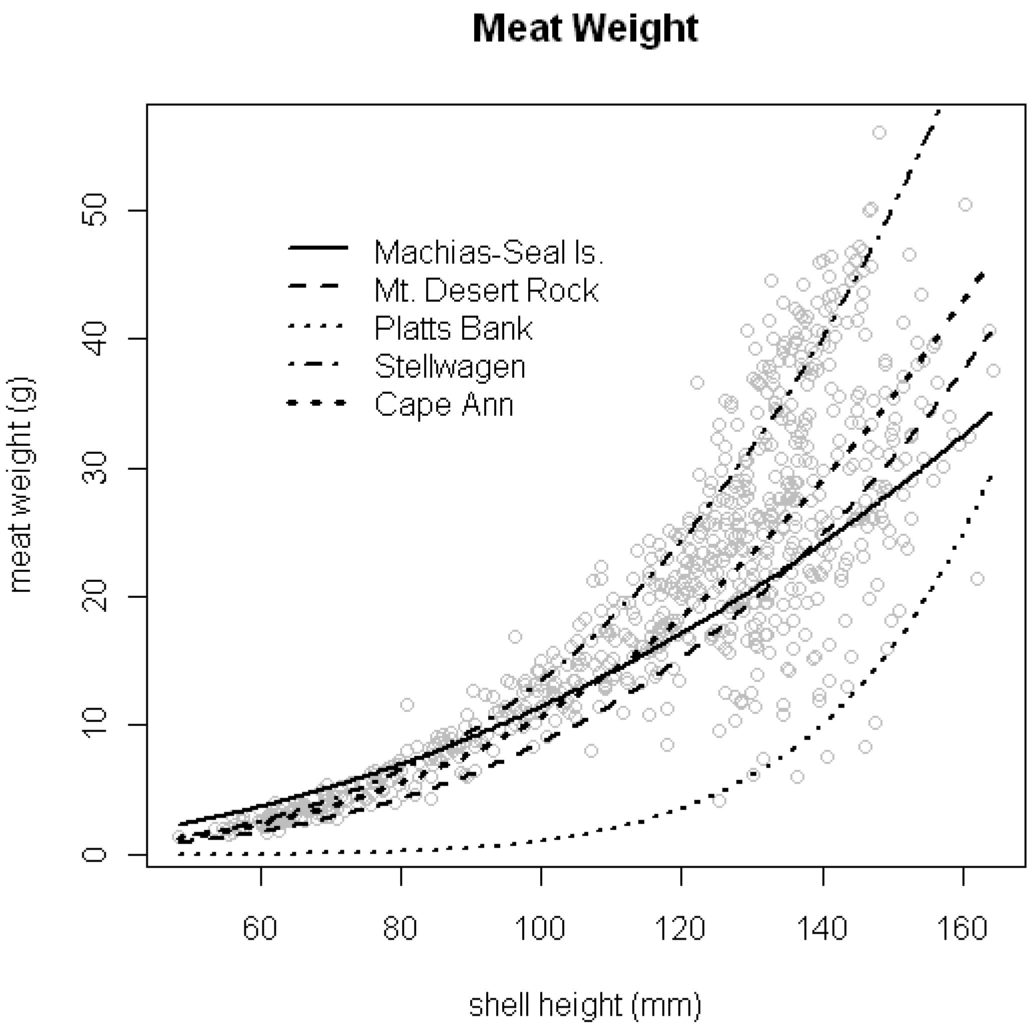
SM4B08
n = 4
SM4A26
n = 2
SM4A28
n =18
SM4C15
n = 2
SM4A23
n =42
SM4A24
n =30
SM4C19
n = 1
SM4C14
n = 2
SM4C18
n = 1
SM4B13
n = 5
proportion
0.0
0.6
50
100
150
shell ht (mm)

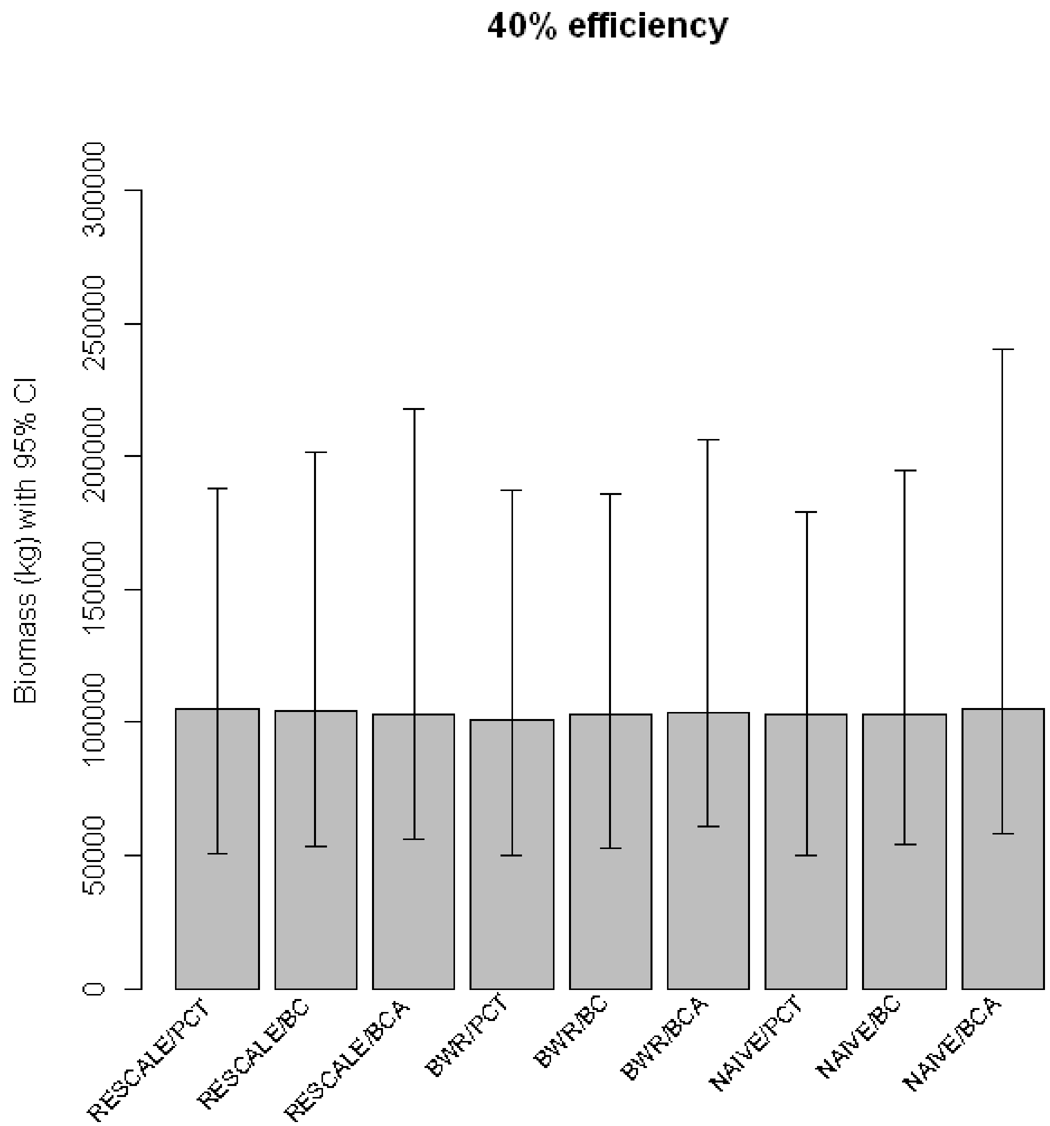

SM5A26
n = 1
SM5A17
n =17
SM5A16
n =14
SM5A24
n = 7
SM5A13
n = 1
SM5A22
n =13
SM5A02
n = 1
SM5A15
n =10
SM5A05
n =18
SM5A12
n =15
SM5A04
n = 1
SM5A21
n =19
SM5A01
n =23
SM5B06
n = 5
SM5B10
n = 2
SM5A29
n = 2
proportion
0.0
0.6
50
100
150
shell ht (mm)



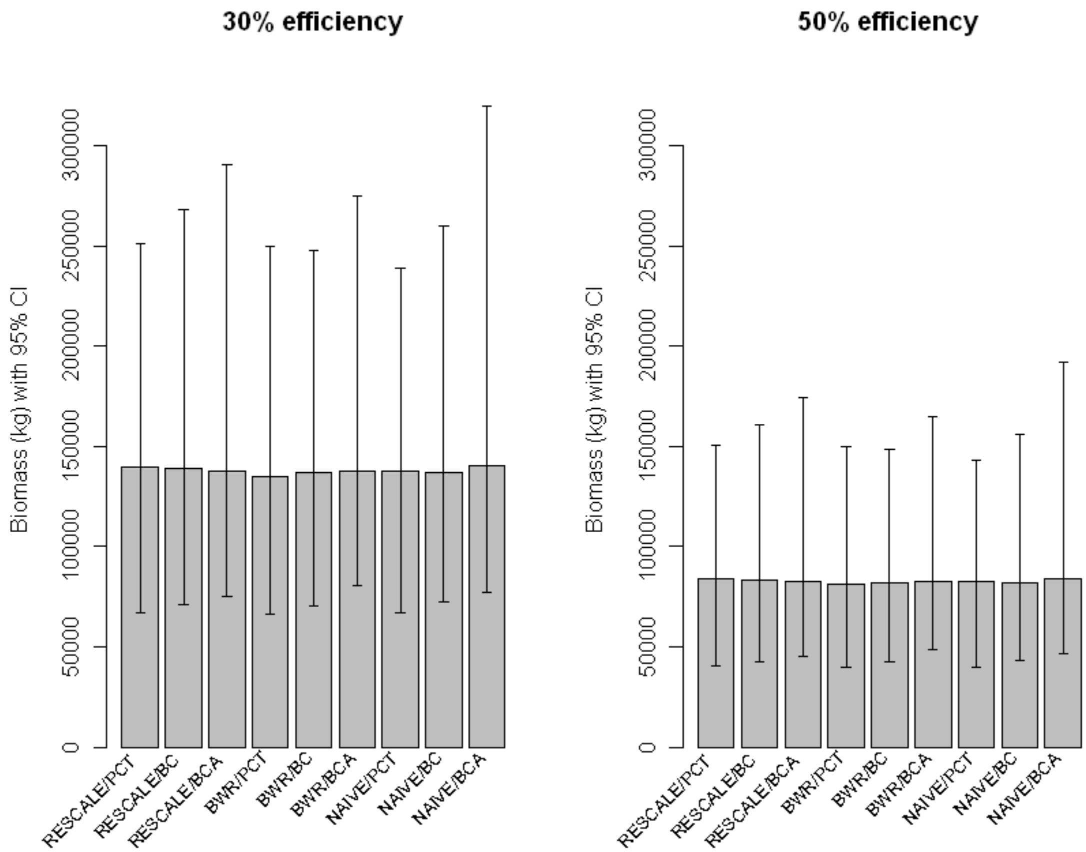
Appendix B6-Figure 4: Individual tow length frequency distributions. Example: SM5A14: 5 represents area 5; A represents subarea A (A is high density, B is medium density, C is low density, D is a tow in state waters); 14 indicates station number.



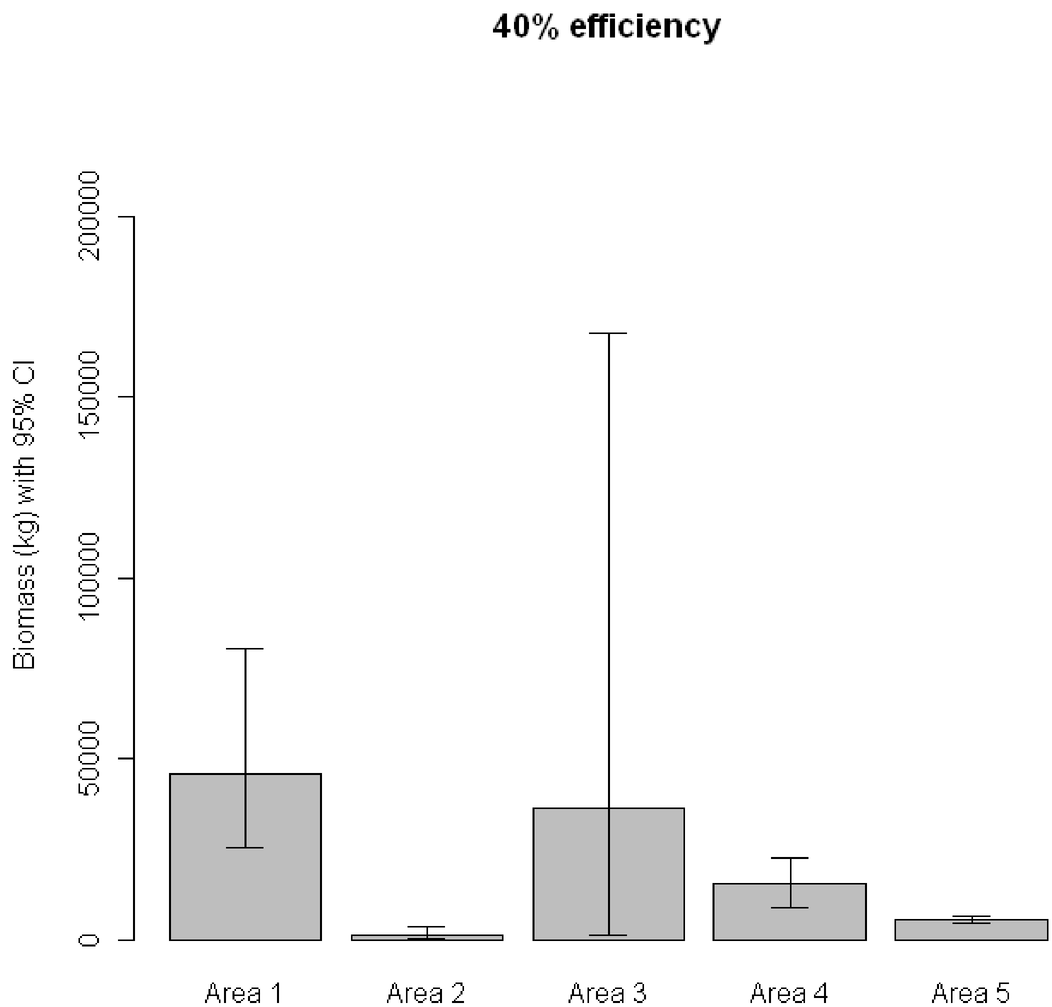
Appendix B6-Figure 5. SH-MW relationship observed for the NGOM survey. The largest meats relative to shell height were found on Stellwagen Bank. The model was $MW_i = \alpha SH_i^{\beta} e^{\varepsilon_i}$. Platts Bank is based on sample size of 8 scallops.



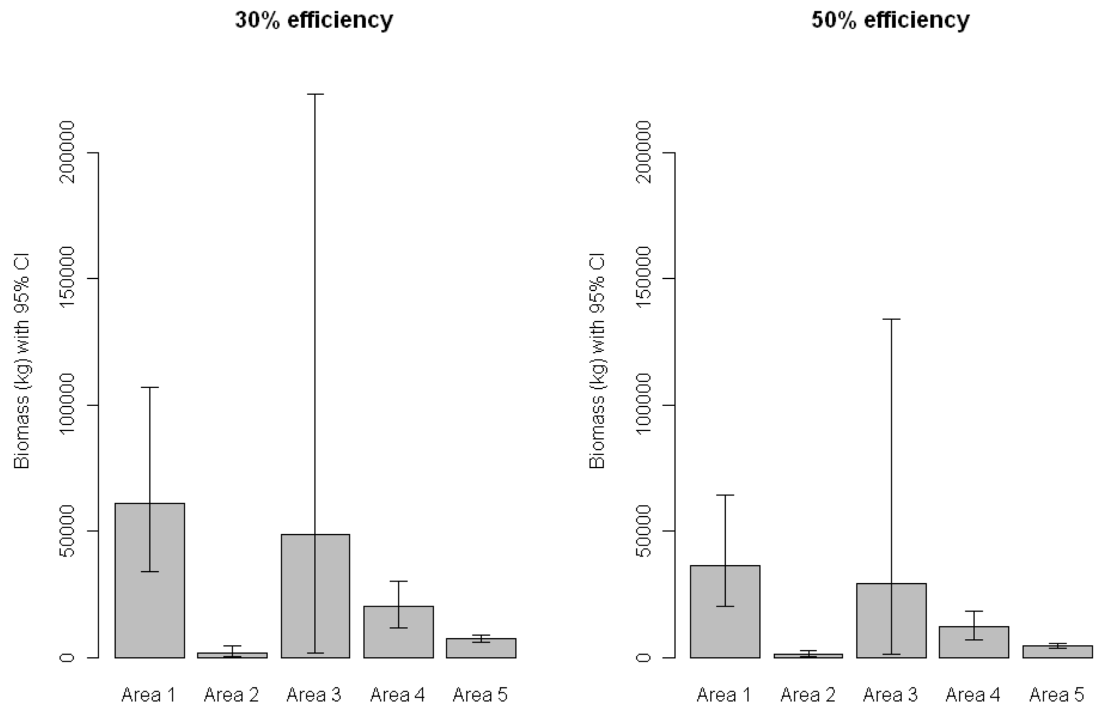
Appendix B6-Figure 6. Mean bootstrapped estimates of NGOM biomass and 95% confidence interval bounds assuming 40% dredge efficiency.



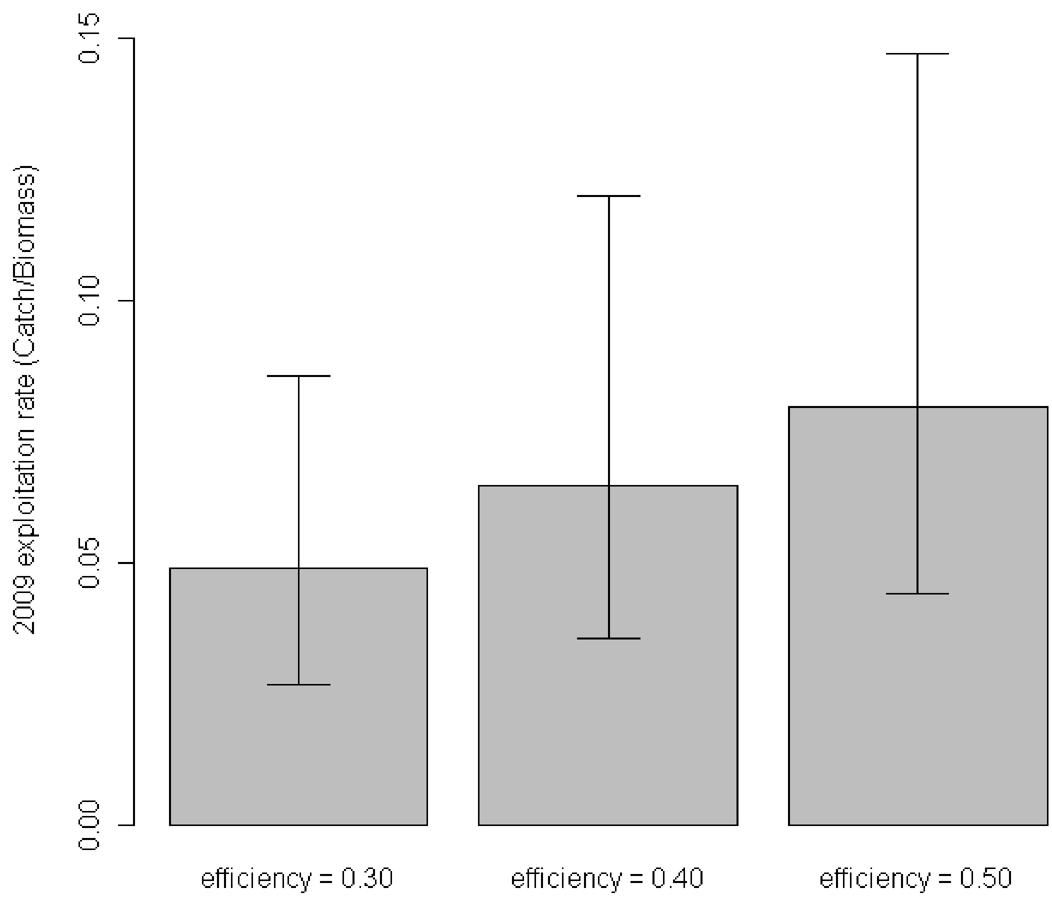
Appendix B6-Figure 7. Mean bootstrapped estimates of NGOM biomass and 95% confidence interval bounds assuming 30% and 50% dredge efficiency.



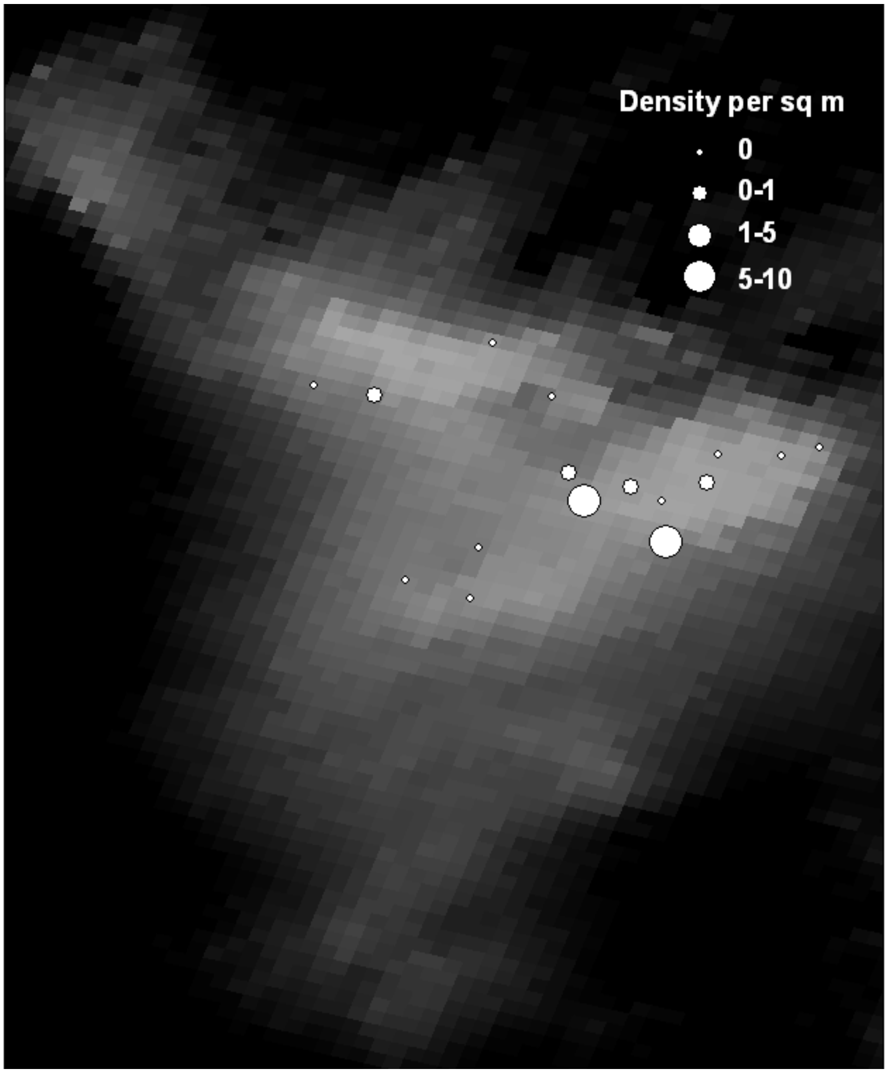
Appendix B6-Figure 8. Mean bootstrapped estimates of NGOM biomass by area and 95% confidence interval bounds using BWR/BC method and assuming 40% dredge efficiency.



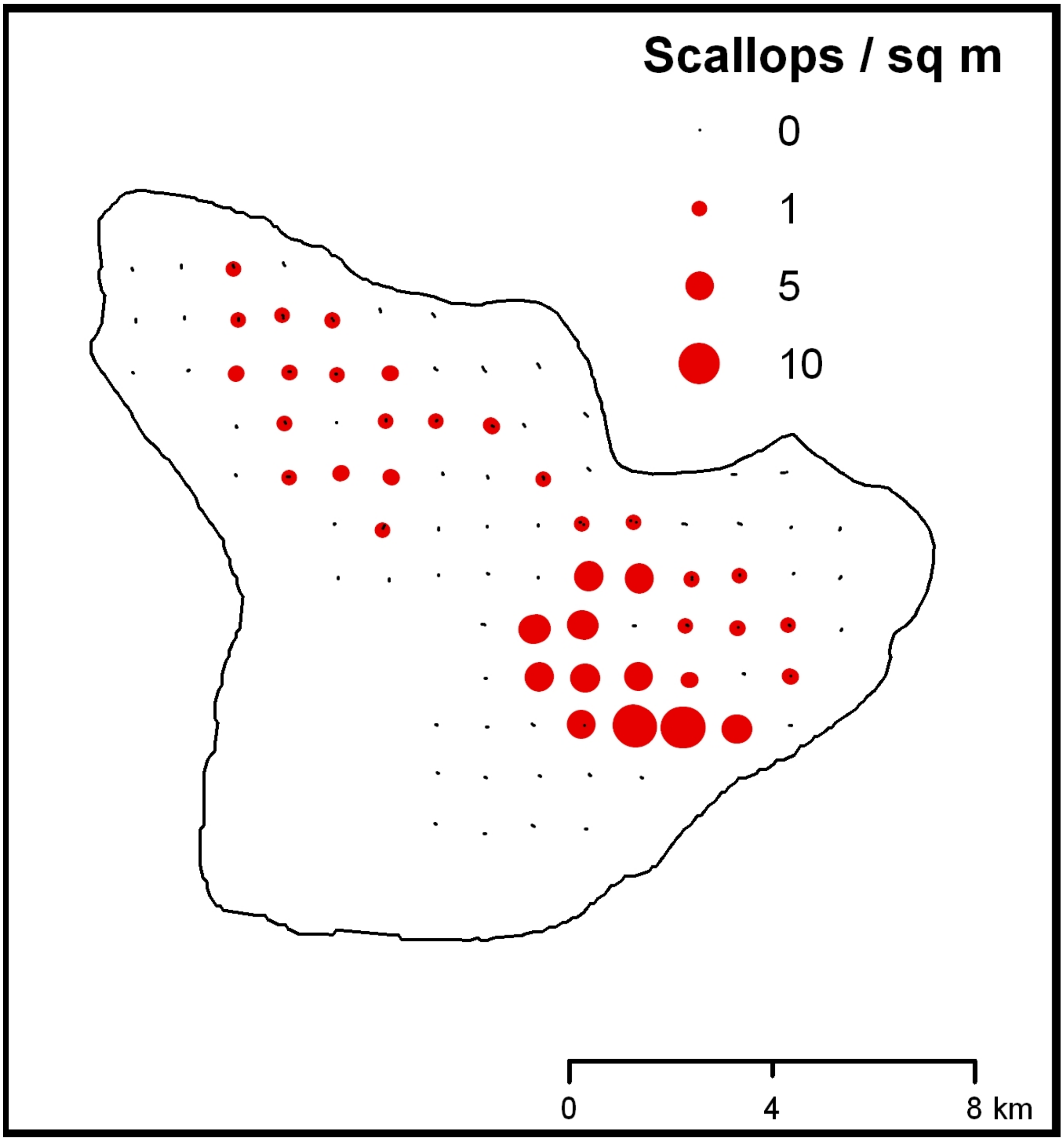
Appendix B6-Figure 9. Mean bootstrapped estimates of NGOM biomass by area and 95% confidence interval bounds using BWR/BC method and assuming 30% and 50% dredge efficiency.



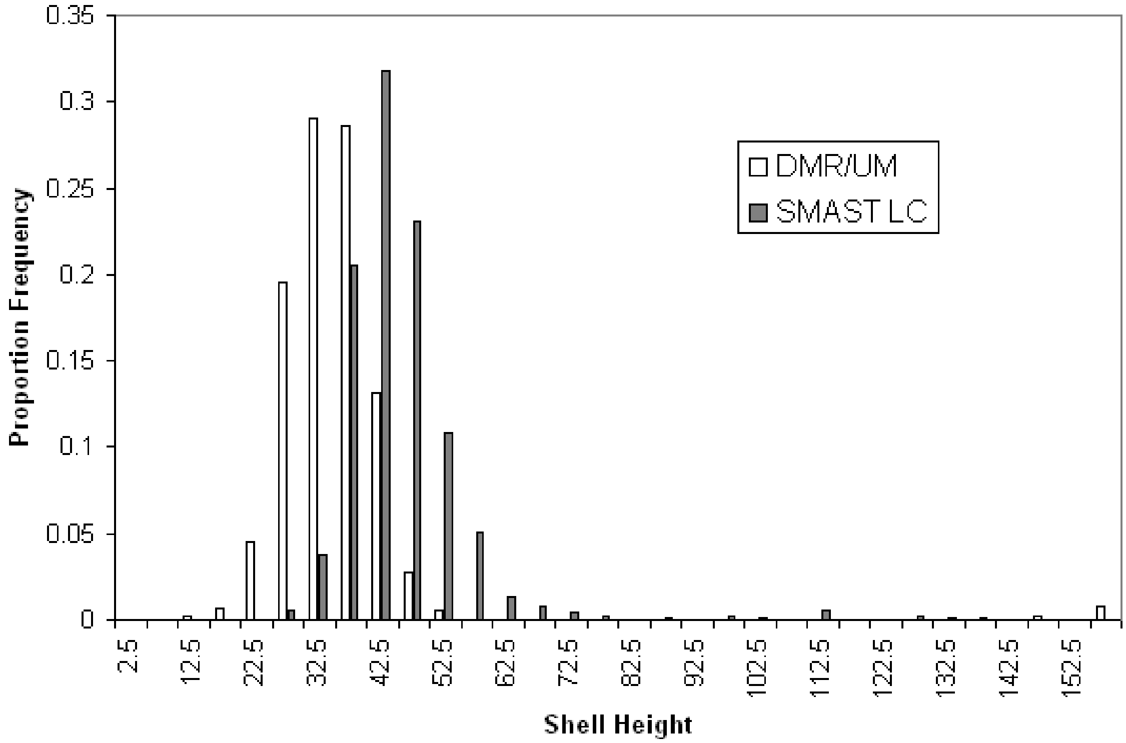
Appendix B6-Figure 10. Estimated NGOM exploitation rates at 30%, 40% and 50% dredge efficiencies with 95% confidence intervals based on BWR/BC method.



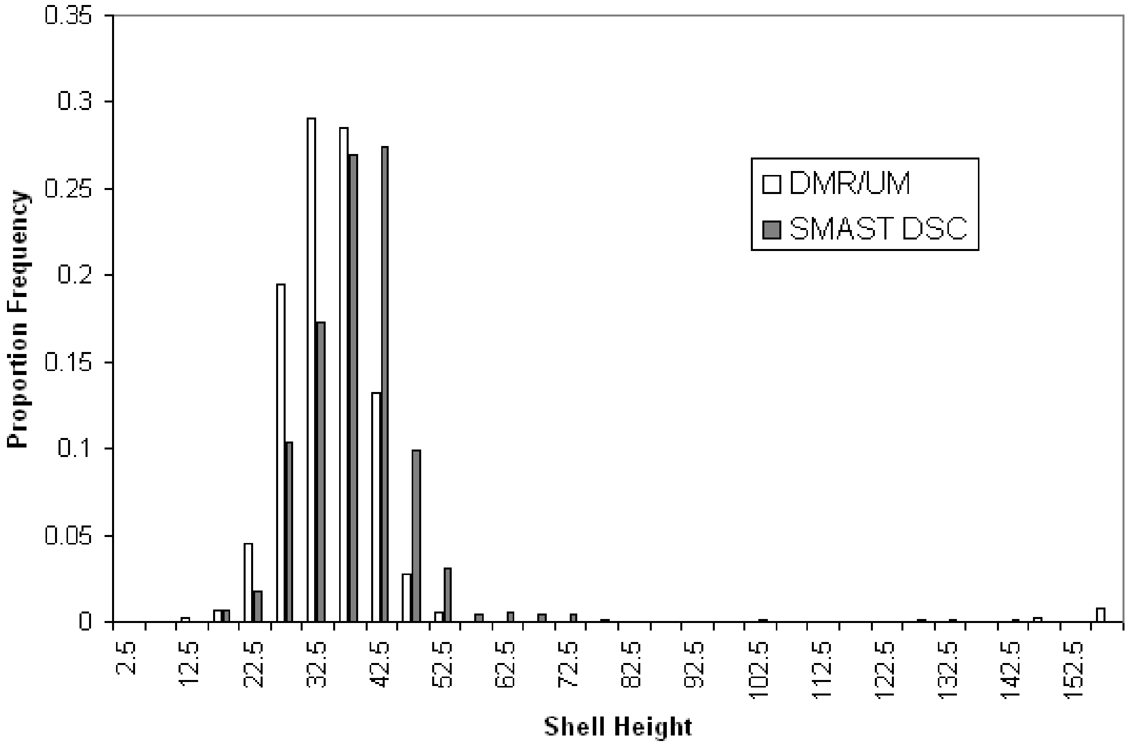
Appendix B6-Figure 11. DMR/UM Platts Bank survey locations indicating density per square meter.



Appendix B6-Figure 12. SMAST Platts Bank survey locations indicating density per square meter.



Appendix B6-Figure 13. Comparison of shell height distribution on Platts Bank between the DMR/UM survey and the SMAST survey (large camera). The DMR survey occurred on July 28th 2009 and the SMAST survey occurred August 12th and 13th 2009.



Appendix B6-Figure 14. Comparison of shell height distribution on Platts Bank between the DMR/UM survey and the SMAST survey (digital still camera). The DMR survey occurred on July 28th 2009 and the SMAST survey occurred August 12th and 13th 2009.

Appendix B7: Shell height-meat weight relationships from NEFSC survey data.

Dan Hennen and Dvora Hart, Northeast Fisheries Science Center, Woods Hole, MA.

New shell height and meat weight data were collected during 2007 – 2009 annual NMFS sea scallop surveys. This appendix updates shell height-meat weight relationships using these data.

Methods

Sea scallops (averaging about 6 per station) were collected for shell height-meat weight analysis at roughly half of all stations during 2001-2009 (717 stations in the Mid-Atlantic, 812 stations on Georges Bank). The scallops were measured to the nearest millimeter, carefully shucked, excess water was removed from the meat, and the meat was weighed to the nearest gram. Samples were collected in 2003, but there was partial data loss, so these data will not be used. During 2001-2009, whole and gonad weights were also recorded, but these data will not be presented here. The sampling protocol was altered slightly in 2009 to begin to account for seasonal shifts in scallop size. Since the data in 2009 were not collected at the same time of year as the data from earlier surveys, 2009 will generally be excluded from this analysis, though it is included in comparisons between years to illustrate the potential effects of shifts in the timing of the survey.

Preliminary analysis indicated a residual pattern for scallops with shell heights less than 70 mm. The small weights of these scallops (1-3 g) combined with the fact that meat weight could only be measured to the nearest gram resulted substantial measurement error. For this reason, the analysis was restricted to scallops that are at least 70 mm shell height. Scallops less than this height are below commercial size and have relatively little influence on CASA model calculations.

A generalized linear mixed model with a log link was used to predict meat weight using shell height, depth, density, latitude, and subarea (a finer scale regional division within each broad region). The GLM used a "quasi" likelihood with a log link, appropriate for data with "constant CV" error (McCullagh and Nelder 1989). This method avoids log-transforming the response variable (meat weight) which can lead to biased estimates when the results are back-transformed. The best model was chosen by AIC (Burnham and Anderson, 2002). The grouping variable for the random effects was a unique code formed by combination of survey station number and the year in which the survey took place. Survey stations were chosen randomly within NEFSC survey strata and generally in proportion to the size of the stratum. Survey stations numbers are assigned sequentially so that a survey station number in one year does not have any particular relationship to the same station number in the next year. Thus, a grouping variable based on a combination of survey station number and year incorporates random variation in the data that is due to both time (year) and fine scale spatial differences (station number).

Several analyses using simplified versions of the best model were employed to explore the effects of year, subarea, and fishing regulations.

All data analysis was conducted using the R statistical program (v2.9.2).

Results

In general, using mixed models appears to be very important in terms of AIC (Table 1). Accounting for the random effects of time and space measured as survey year and location absorbs much of the variation in the data.

Mid-Atlantic
The following model had the lowest AIC value (Table 1).

$$W = e^{\left(\alpha + a(St) + \beta \ln(H) + \gamma ln(D) + \rho(\ln(L) * \ln(D))\right) + \epsilon} \quad (1)$$

Where *W* was meat weight (g), and St was the year-station grouping variable for the random effects. The random effects were always modeled as an intercept and sometimes as a slope coefficient. The fixed effects were: shell height (*H)* in mm, depth (*D*) in m, and an interaction between shell height and depth (*H**D*). A total of 4181 observations were sampled from 717 stations were used in the analysis (Figure 1). Parameters (Table 1) were well estimated with no evidence of residual patterns (Table 2, Figure 2-4). The estimates presented here were similar to most previous estimates (Table 3). Compared to the estimates used in previous assessments, with the exception of Lai and Helser (2004) (Figure 5), the new estimates predicted slightly heavier meats at small shell heights, but lighter meats at very large shell heights, though the differences were small. The relationship that includes a depth effect indicated that sea scallops have heavier meats at shallower depths (Figure 6).

Meat weights varied by year, with the heaviest meats during 2004 (Table 4, Figure 7). Meats were generally heavier in 2009 when the survey was conducted earlier in the year. Meat weights by subarea were less variable, though "New York Bight" did produce heavier meats at the larger shell heights, and particularly at deeper depths (60 and 70 m) than the other areas (Table 5, Figures 8-10). In general samples taken from the Mid-Atlantic tend to be from water shallower than 70 m (Figure 11).

Georges Bank
The following model had the lowest AIC value (Table 1).

$$W = e^{(\alpha + a(St) + \beta ln(H) + \gamma ln(D) + \delta ln(lat) + \theta(sub) + b(L_{St})) + \epsilon} \quad (2)$$

Where *W* was meat weight (g), and survey station (St) was the grouping for the random effects. The random effects were modeled as an intercept (*a*), and as a slope parameter (*b*) for shell height (*H*). The fixed effects were in mm, *D* in m, latitude (*lat*) in decimal degrees, and subarea (*sub*) based on area management boundaries. Based on 6145 scallops from 812 stations, model fits appeared good with little or no residual pattern (Figures 12-15). Parameters were reasonably precise (Tables 1 and 2). They predict slightly heavier meat weights at small shell heights, and slightly lighter meat weights at large shell heights, than the model used in the previous assessment (Table 3, Figure 16). Meat weights were heavier at shallower depths (Figure 17).

Scallop shell height-meat weight relationships were generally consistent over time, although recent years (2007 and 2008) had heavier meats for large shell heights (Table 4, Figure 18). The 2009 survey which was conducted earlier in the year than previous surveys, collected meats that tended to be heavier at small shell heights, but did not otherwise differ from meats collected in other years. Results were dependent on subareas with "South East Part" and

“Closed Area 1” producing larger meats and “South Channel” and “Northern Edge and Peak” tending to produce lighter meats at all shell heights at the shallower depths (50 and 60 m) (Table 5, Figures 19 - 20). At 90 m depth, the heaviest meats were found in Northern Light Ship area at all shell heights and South East Channel produced some of the smallest meats at all shell heights. It should be noted however that samples from Northern Light Ship area were all taken from waters less than 90 m deep so the heavy weights found by the model fit could be an artifact of sampling (Figure 21). Areas that were closed to fishing tended to have larger meats (Figure 23).

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Appendix B7-Table 1. Model building results. The models with minimum AIC values are indicated by bold font. Random effects are shown as parameters inside parentheses. All random effects were grouped by year_station and each model included a random intercept represented by the 1 inside parentheses. Fixed effects are shown to the right of the ~ symbol which separates the response variable from the predictors. Interaction terms are represented as factor1 * factor2. The best model tested without random effects for each region is included for comparison.

Formula	AIC	BIC	logLik	deviance
Georges Bank				
meat_weight ~ height + depth + lat + subarea + (height + 1 | year_station)	6636	6723	-3305	6610
meat_weight ~ height + depth + subarea + (height + 1 | year_station)	6694	6774	-3335	6670
meat_weight ~ height + depth + subarea + height * depth + (height + 1 | year_station)	6696	6783	-3335	6670
meat_weight ~ height + subarea + height * depth + (height + 1 | year_station)	6696	6783	-3335	6670
meat_weight ~ height + depth + lat + (height + 1 | year_station)	6707	6761	-3346	6691
meat_weight ~ height + depth + density + lat + (height + 1 | year_station)	6708	6769	-3345	6690
meat_weight ~ height + depth + lat + height * depth + (height + 1 | year_station)	6709	6770	-3346	6691
meat_weight ~ height + depth + lat + (1 | year_station)	6761	6801	-3374	6749
meat_weight ~ height + depth + subarea + (1 | year_station)	6761	6828	-3370	6741
meat_weight ~ height + depth + density + lat + (1 | year_station)	6762	6809	-3374	6748
meat_weight ~ height + depth + (height + 1 | year_station)	6786	6833	-3386	6772
meat_weight ~ height + depth + density + (height + 1 | year_station)	6788	6841	-3386	6772
meat_weight ~ depth + height * depth + (height + 1 | year_station)	6788	6842	-3386	6772
meat_weight ~ height + height * depth + (height + 1 | year_station)	6788	6842	-3386	6772
meat_weight ~ height + density + height * depth + (height + 1 | year_station)	6790	6850	-3386	6772
meat_weight ~ height + depth + density + height * depth + (height + 1 | year_station)	6790	6850	-3386	6772
meat_weight ~ height + depth + (1 | year_station)	6839	6873	-3414	6829
meat_weight ~ height + depth + density + (1 | year_station)	6840	6881	-3414	6828
meat_weight ~ depth + height * depth + (1 | year_station)	6841	6881	-3414	6829
meat_weight ~ height + height * depth + (1 | year_station)	6841	6881	-3414	6829
meat_weight ~ height + lat + (height + 1 | year_station)	6988	7035	-3487	6974
meat_weight ~ height + (height + 1 | year_station)	7040	7081	-3514	7028
meat_weight ~ height + lat + (1 | year_station)	7041	7075	-3515	7031
meat_weight ~ height + density + (height + 1 | year_station)	7042	7090	-3514	7028
meat_weight ~ height + (1 | year_station)	7093	7120	-3542	7085
meat_weight ~ height + density + (1 | year_station)	7095	7128	-3542	7085
meat_weight ~ depth + (height + 1 | year_station)	9074	9115	-4531	9062
meat_weight ~ depth + (1 | year_station)	29295	29322	-14643	29287
meat_weight ~ height + depth + height * depth + lat + subarea	42747		-6107	376871

Mid-Atlantic Bight				
*meat_weight ~ depth + height * depth + (1 | year_station)*	3626	3664	-1807	3614
meat_weight ~ height + height * depth + (1 | year_station)	3626	3664	-1807	3614
meat_weight ~ height + depth + density + height * depth + (height + 1 | year_station)	3629	3686	-1806	3611
meat_weight ~ height + density + height * depth + (height + 1 | year_station)	3629	3686	-1806	3611
meat_weight ~ height + height * depth + (height + 1 | year_station)	3630	3681	-1807	3614
meat_weight ~ depth + height * depth + (height + 1 | year_station)	3630	3681	-1807	3614
meat_weight ~ height + depth + lat + height * depth + (height + 1 | year_station)	3631	3688	-1807	3613
meat_weight ~ height + depth + subarea + height * depth + (height + 1 | year_station)	3632	3708	-1804	3608
meat_weight ~ height + subarea + height * depth + (height + 1 | year_station)	3632	3708	-1804	3608
meat_weight ~ height + depth + density + (1 | year_station)	3634	3672	-1811	3622
meat_weight ~ height + depth + (1 | year_station)	3635	3667	-1813	3625
meat_weight ~ height + depth + subarea + (1 | year_station)	3636	3693	-1809	3618
meat_weight ~ height + depth + density + lat + (1 | year_station)	3636	3681	-1811	3622
meat_weight ~ height + depth + density + (height + 1 | year_station)	3637	3687	-1810	3621
meat_weight ~ height + depth + lat + (1 | year_station)	3637	3675	-1812	3625
meat_weight ~ height + depth + (height + 1 | year_station)	3638	3682	-1812	3624
meat_weight ~ height + depth + subarea + (height + 1 | year_station)	3638	3708	-1808	3616
meat_weight ~ height + depth + density + lat + (height + 1 | year_station)	3638	3696	-1810	3620
meat_weight ~ height + depth + lat + (height + 1 | year_station)	3639	3690	-1812	3623
meat_weight ~ height + depth + lat + subarea + (height + 1 | year_station)	3640	3716	-1808	3616
meat_weight ~ height + lat + (1 | year_station)	3838	3870	-1914	3828
meat_weight ~ height + lat + (height + 1 | year_station)	3841	3886	-1914	3827
meat_weight ~ height + (1 | year_station)	3848	3873	-1920	3840
meat_weight ~ height + density + (1 | year_station)	3848	3880	-1919	3838
meat_weight ~ height + (height + 1 | year_station)	3851	3889	-1919	3839
meat_weight ~ height + density + (height + 1 | year_station)	3851	3895	-1918	3837
meat_weight ~ depth + (height + 1 | year_station)	5644	5682	-2816	5632
meat_weight ~ depth + (1 | year_station)	15340	15365	-7666	15332
meat_weight ~ height + depth	26144		-8715	126965

Appendix B7-Table 2. The standard errors for the parameter estimates in Table 1. The parameters estimated are: the intercept (α), the shell height coefficient (β), the depth coefficient (γ), the latitude coefficient (δ), and (ρ) the shell height by depth interaction in MAB, and the subarea coefficient in GBK.

	α	β	γ	δ	ρ	resid.
Mid-Atlantic Bight						
NEFSC (2007)	0.150	0.050				
NEFSC (2007) with Depth effect	0.390	0.050	0.080			
NEFSC (2010)	0.024	0.096				3.61[a]
NEFSC (2010) with Depth effect	0.021	0.093	0.104			3.61[a]
NEFSC (2010) with Depth effect and interaction	0.021	0.095	0.106		0.472	3.61[a]
Georges Bank						
NEFSC (2007)	0.270	0.060				
NEFSC (2007) with Depth effect	0.170	0.050	0.050			
NEFSC (2010)	0.034	0.090				4.57[a]
NEFSC (2010) with Depth and Latitude effect	0.028	0.102	0.131			4.46[a]
NEFSC (2010) with Depth, Latitude and subarea effect	0.061	0.104	0.129	3.286	0.098[b]	4.46[a]

a - these are standard deviations

b - averaged across all subarea levels

Appendix B7-Table 3. Current shell height/meat weight parameters, compared with those from other studies. The parameters estimated are: the intercept (α), the shell height coefficient (β), the depth coefficient (γ); the latitude coefficient (δ), and (ρ) the shell height by depth interaction in MAB, and the average subarea coefficient in GBK.

	α	β	γ	δ	ρ
Mid-Atlantic Bight					
Haynes (1966)	-11.09	3.04			
Serchuk and Rak (1983)	-12.16	3.25			
NEFSC (2001)	-12.25	3.26			
Lai and Helser (2004)	-12.34	3.28			
NEFSC (2007)	-12.01	3.22			
NEFSC (2007) with Depth effect	-9.18	3.18	-0.65		
NEFSC (2010)	-10.80	2.97			
NEFSC (2010) with Depth effect	-8.94	2.94	-0.43		
NEFSC (2010) with Depth effect and interaction	-16.88	4.64	1.57	-	-0.43
Georges Bank					
Haynes (1966)	-10.84	2.95			
Serchuk and Rak (1983)	-11.77	3.17			
NEFSC (2001)	-11.60	3.12			
Lai and Helser (2004)	-11.44	3.07			
NEFSC (2007)	-10.70	2.94			
NEFSC (2007) with Depth effect	-8.62	2.95	-0.51		
NEFSC (2010)	-10.25	2.85			
NEFSC (2010) with Depth effect	-8.05	2.84	-0.51		
NEFSC (2010) with Depth, Latitude and subarea effect	14.380	2.826	-0.529	5.980	0.051[b]

b - averaged across all subarea levels

Appendix B7-Table 4. Current shell height/meat weight parameters, compared across years. The parameters estimated are: the intercept (α), the shell height coefficient (β), the depth coefficient (γ). The numbers of stations used in each year are also shown.

	α	β	γ	$n_{(stations)}$
Mid-Atlantic Bight[b]				
2001	-10.40	2.97	-0.1007	69
2002	-8.54	2.86	-0.4601	54
2003[a]				
2004	-9.70	2.98	-0.2592	124
2005	-8.60	3.12	-0.7516	130
2006	-8.75	3.05	-0.6331	111
2007	-8.83	2.77	-0.2365	120
2008	-8.03	2.80	-0.4744	109
2009	-8.44	2.75	-0.303	101
Georges Bank[b]				
2001	-7.7695	2.8203	-0.5614	52
2002	-7.3727	2.72	-0.5394	90
2003[a]				
2004	-7.9818	2.7536	-0.4313	154
2005	-8.3563	2.8691	-0.477	137
2006	-7.0069	2.728	-0.6328	135
2007	-7.6659	2.9681	-0.7194	155
2008	-9.247	2.9165	-0.3091	89
2009	-7.1515	2.5507	-0.3874	110

a - estimates using 2003 survey data were excluded from the model

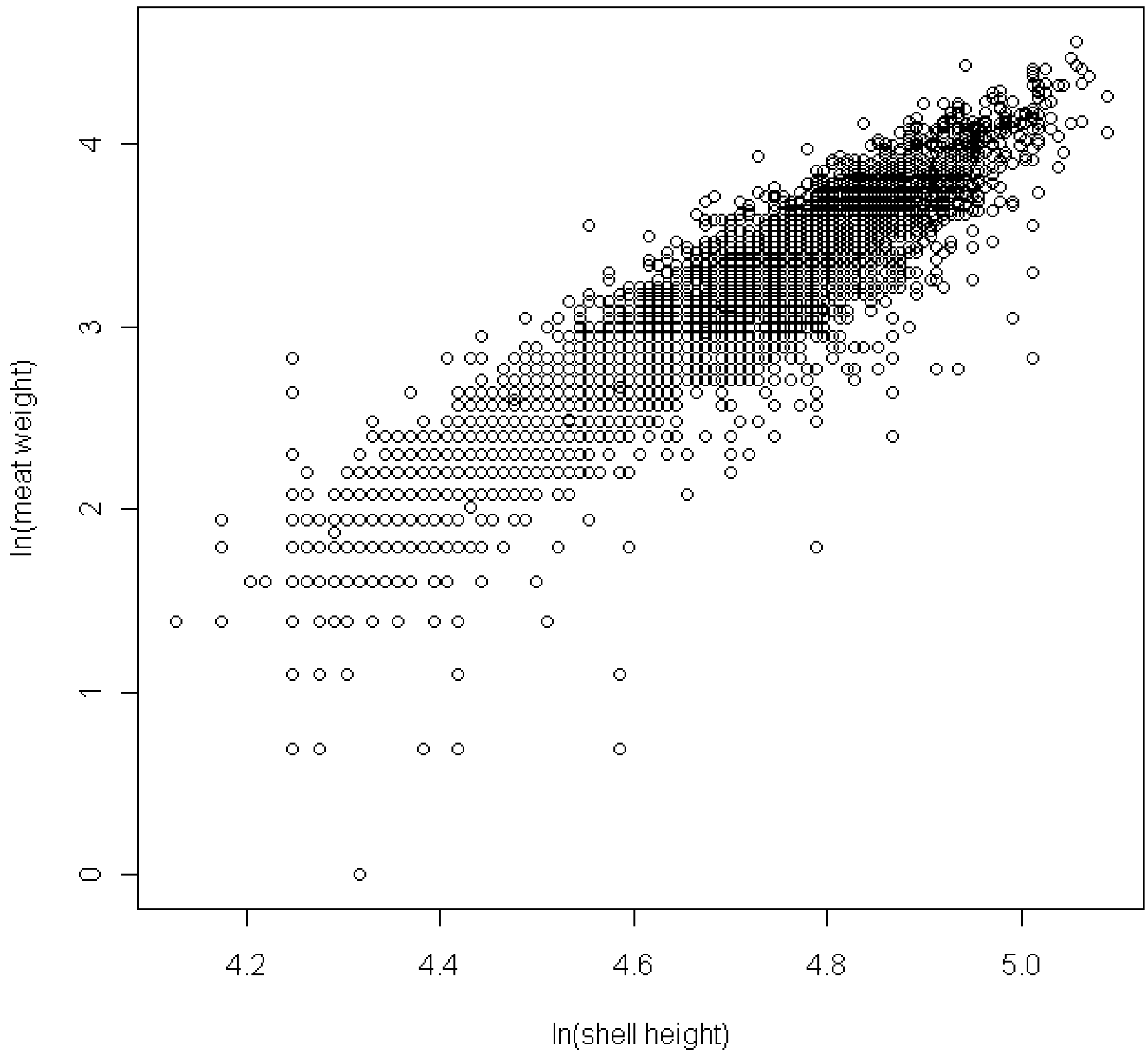
b - model = meat_weight ~ height + depth + (1 | year_station)

Appendix B7-Table 5. Current shell height/meat weight parameters, compared across subareas within each region. The parameters estimated are: the intercept (α), the shell height coefficient (β), the depth coefficient (γ). The numbers of stations used in each year are also shown.

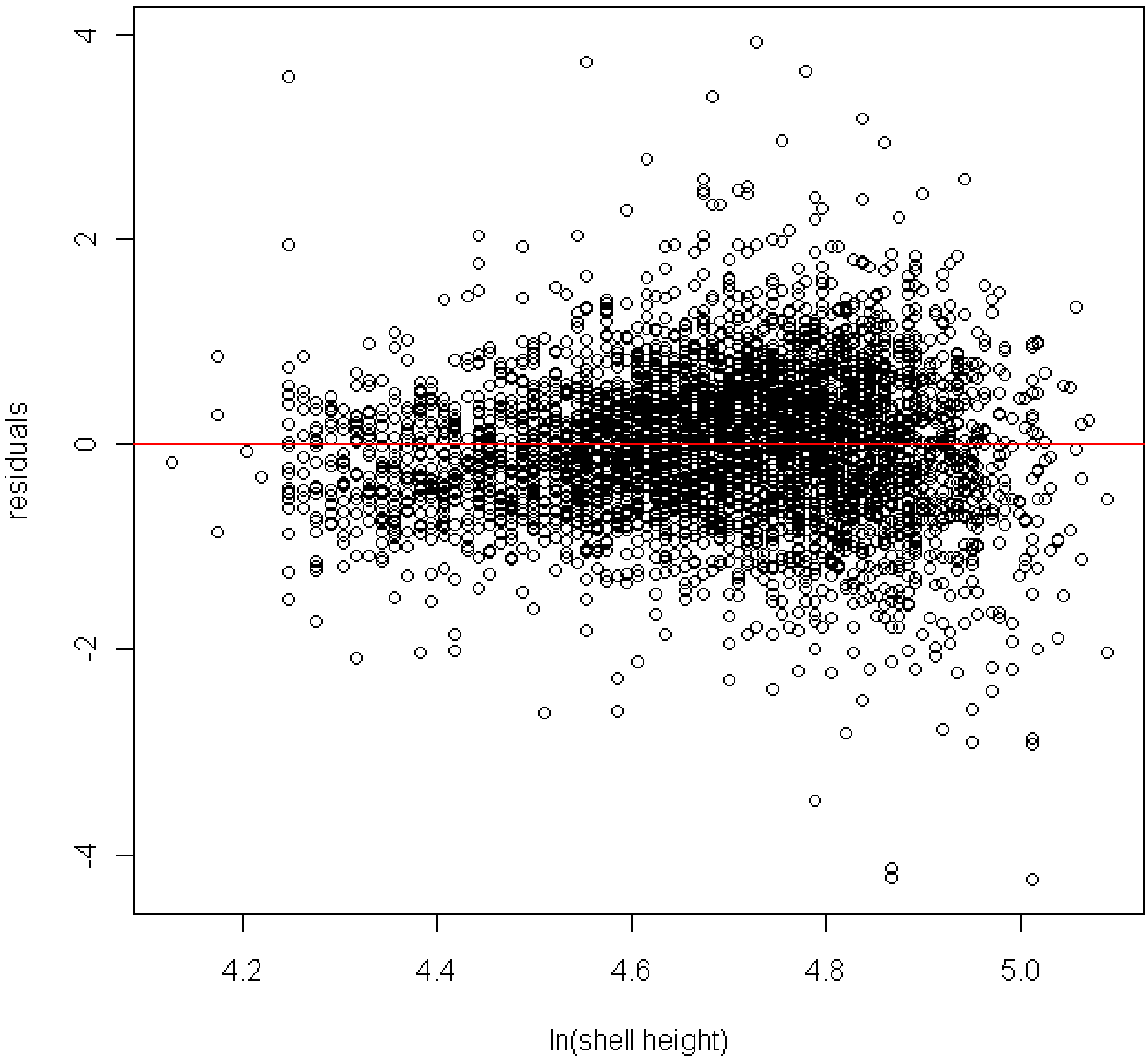
	α	β	γ	$n_{(stations)}$
Mid-Atlantic Bight[a]				
DMV-VB	-8.0407	2.8249	-0.5194	125
ET	-7.0358	2.9036	-0.861	194
HC	-7.305	2.9066	-0.7863	139
LI	-9.7815	2.9439	-0.224	150
NYB	10.3701	3.0698	-0.213	109
Georges Bank[b]				
CL-1	-6.3757	2.7999	-0.8405	148
CL-2	-8.7026	2.8338	-0.3354	205
NEP	-7.9355	2.8325	-0.5477	152
NLS	-8.1709	2.6454	-0.2298	92
Sch	-9.5245	2.9359	-0.2808	146
SEP	-4.3756	2.6291	-1.1166	69

a - model = meat_weight ~ height + depth + (1 | year_station)

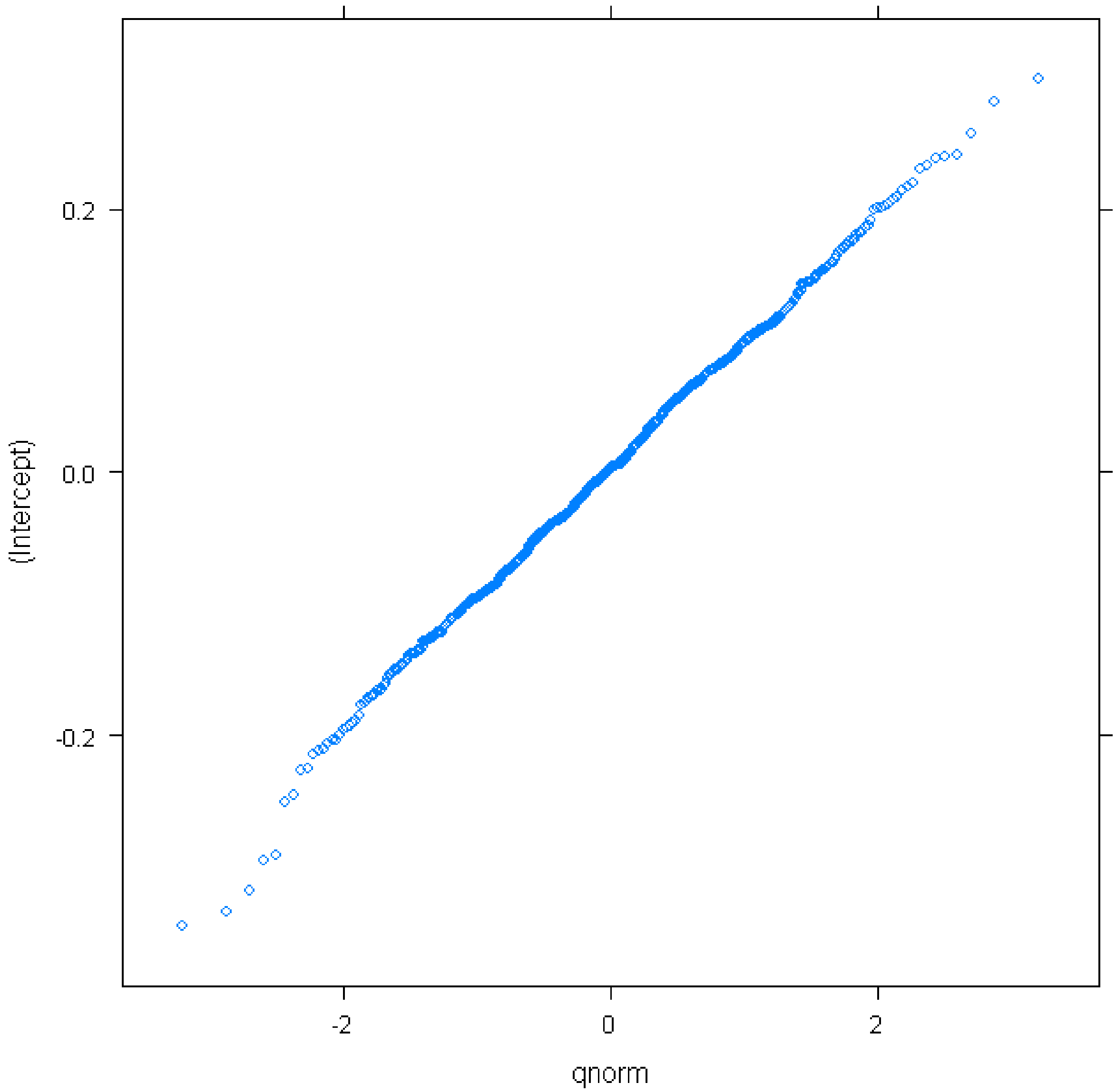
b - model = meat_weight ~ height + depth + (height + 1 | year_station)



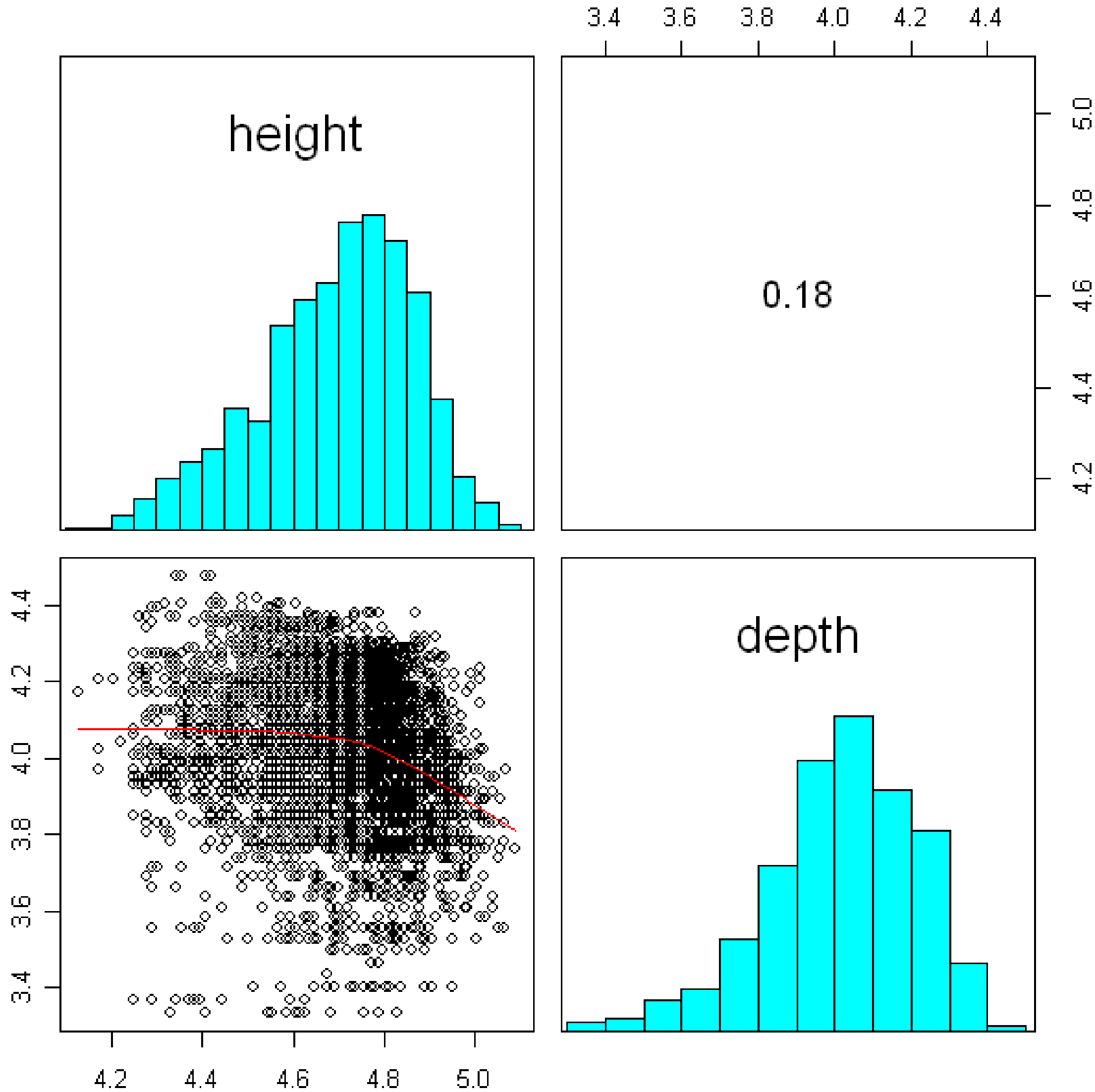
Appendix B7-Figure 1. Mid-Atlantic shell height/meat weight data



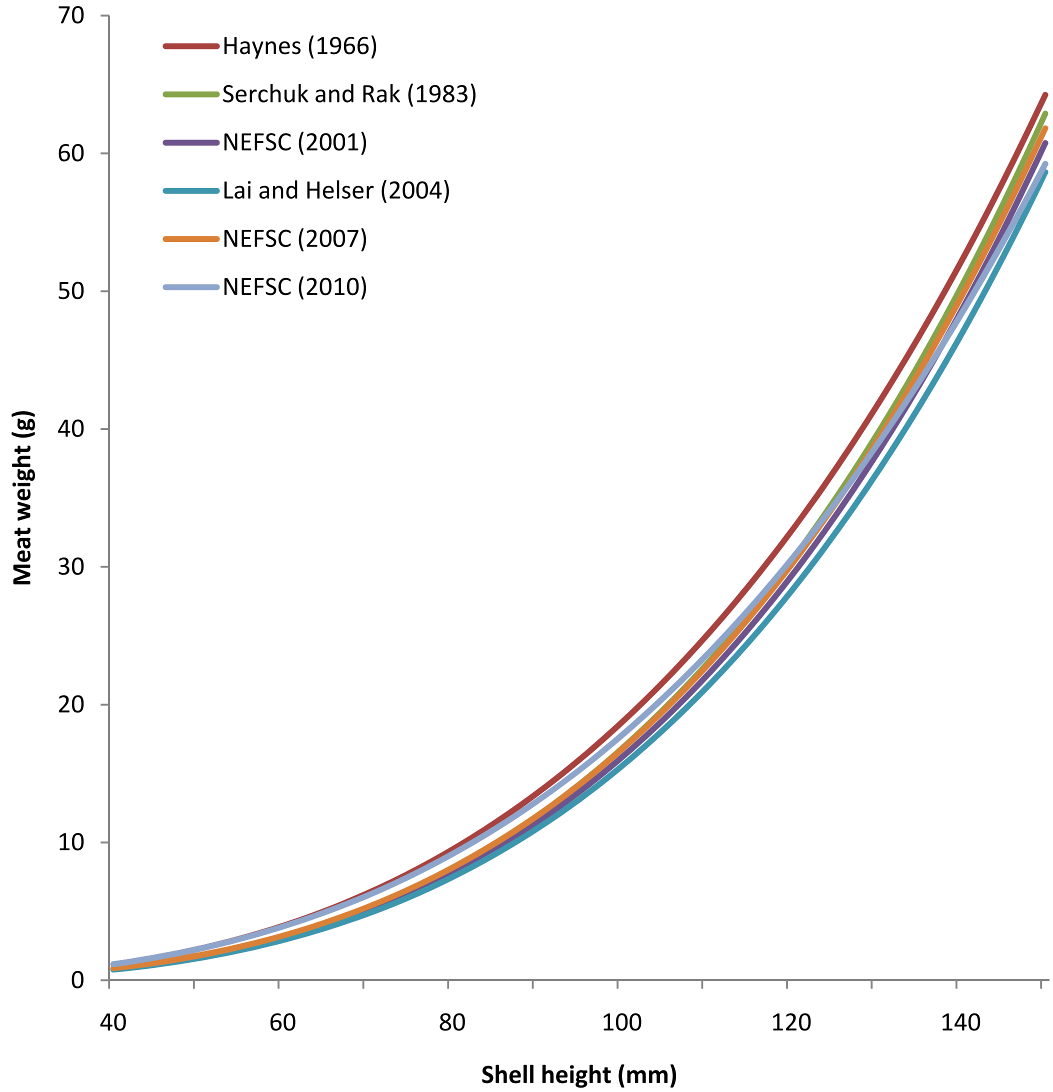
Appendix B7-Figure 2. Residual plot of Mid-Atlantic shell height/meat weight data



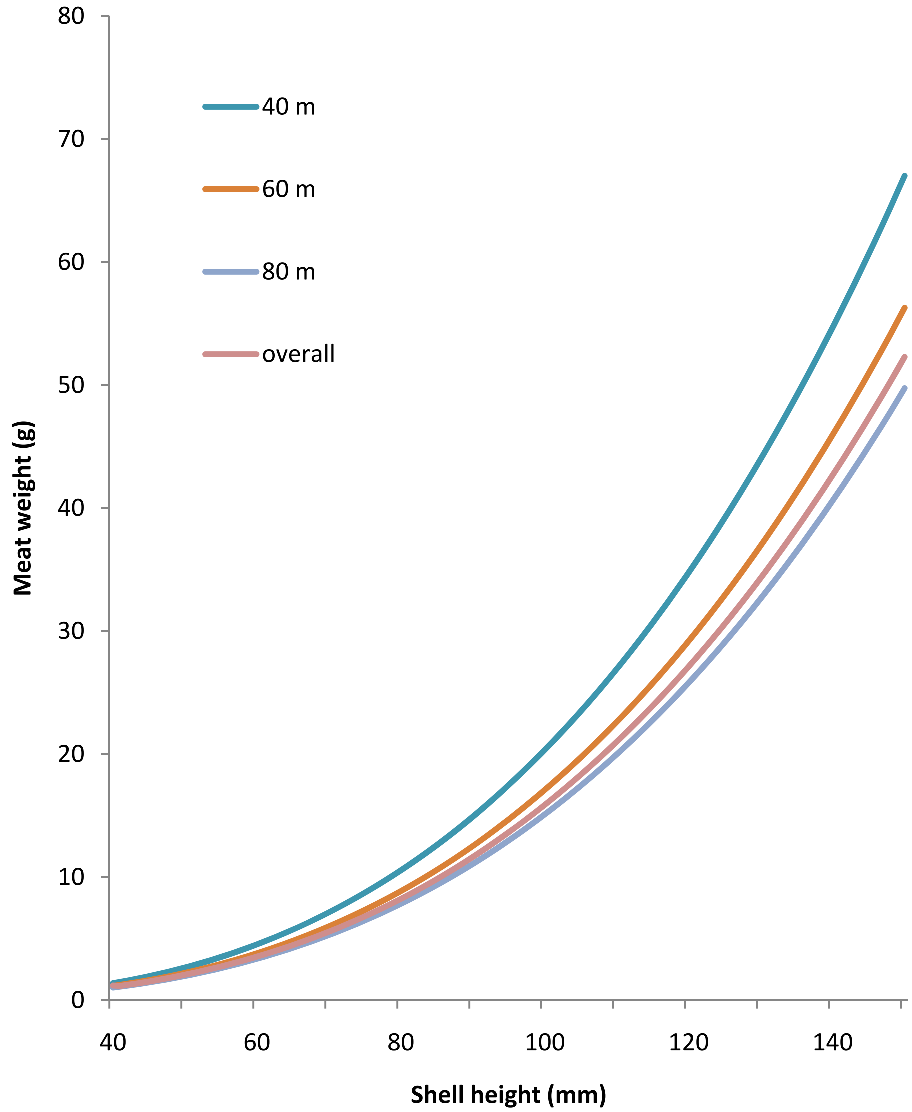
Appendix B7-Figure 3. Normality plot of the BLUPs (Best Linear Unbiased Predictions of the random effects) from the best model (Eq. 1) for the Mid-Atlantic Bight. The only random effect is an intercept, grouped by station (where station is a unique identifier that incorporates spatial – survey station, and temporal – year, variability).



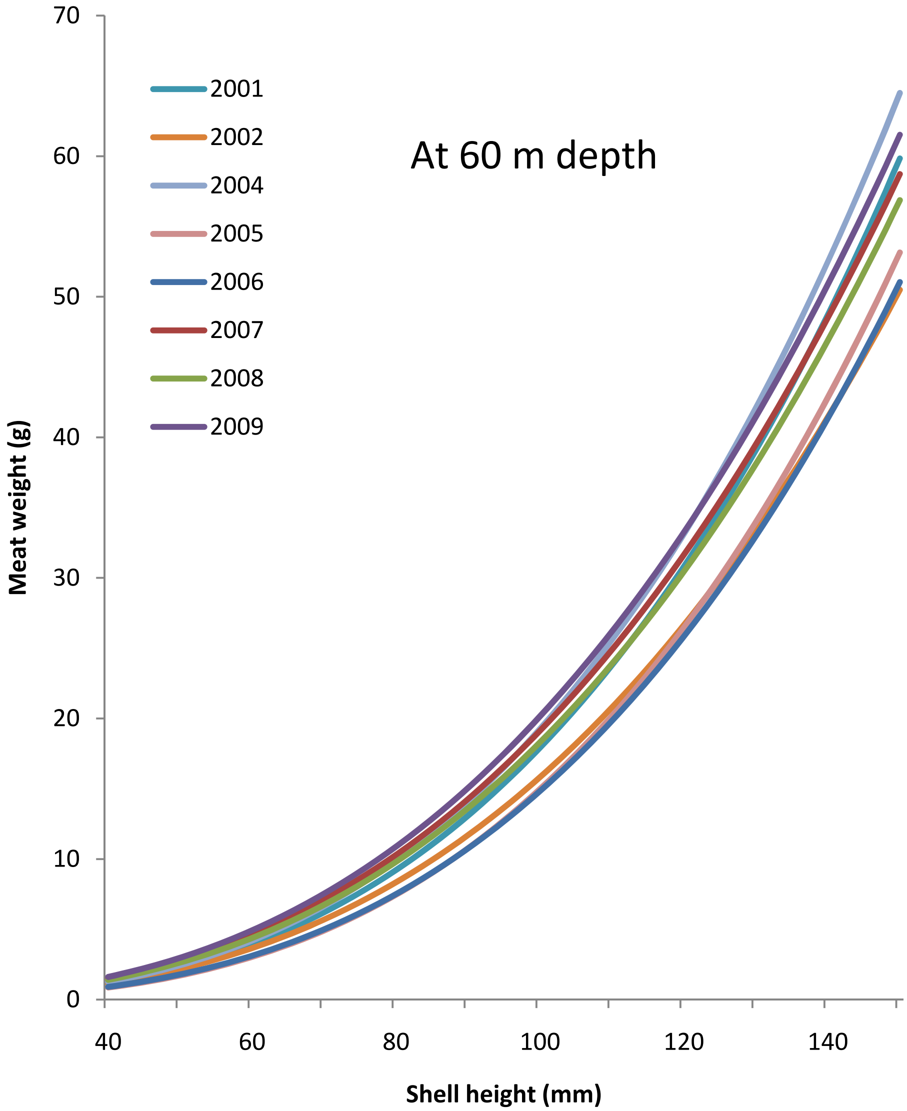
Appendix B7-Figure 4. The correlation plot of the fixed effects from the best model (Eq. 1) for the Mid-Atlantic Bight. The values of the correlation coefficients for each comparison are shown in the upper diagonal. The main diagonal shows the frequency histogram of each effect and the scatter plot in the lower diagonal includes a smooth curve meant only to aid visual interpretation.



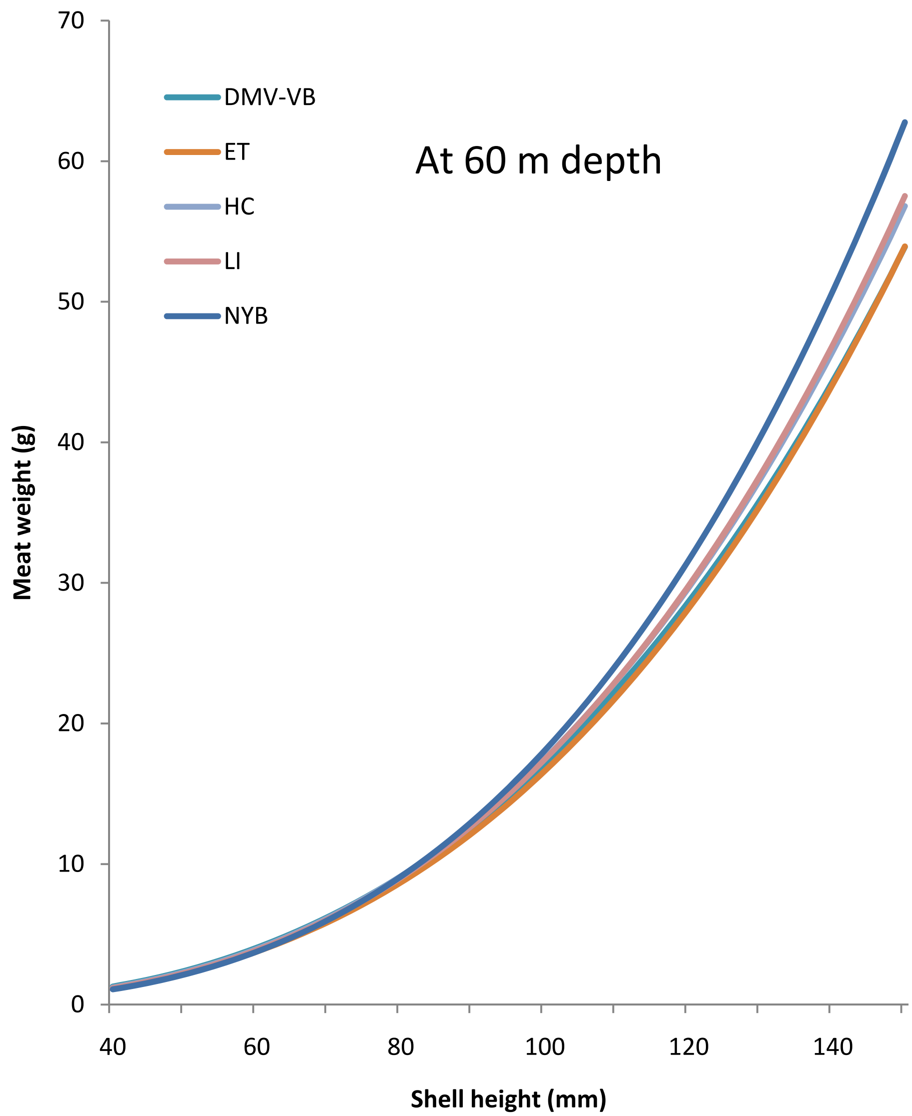
Appendix B7-Figure 5. Comparison of historical shell height/meat weight parameter estimates in the Mid-Atlantic (directly comparable models only, i.e. of the form $\boldsymbol{W} = \boldsymbol{e}^{(\alpha+a(St)+\beta)+\epsilon}$).



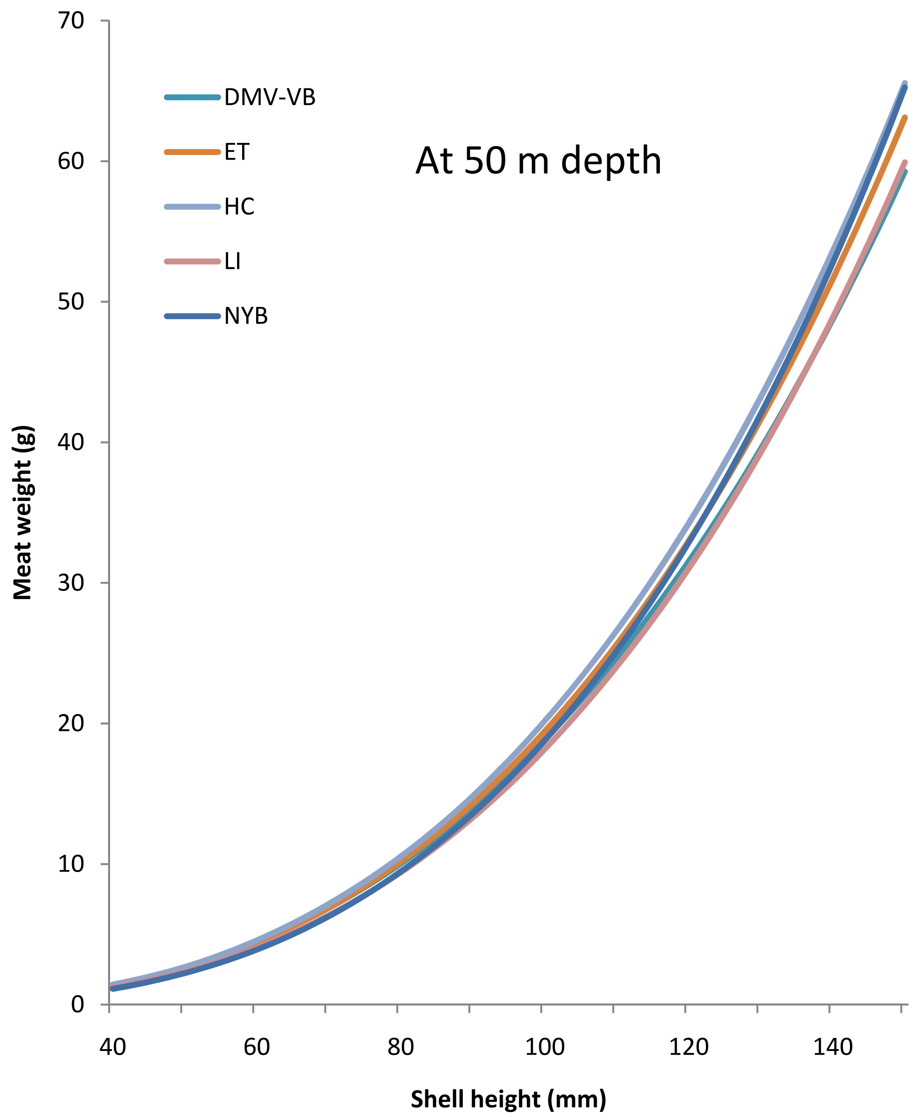
Appendix B7-Figure 6. Shell height/meat weight relationships at relationships 40, 60, 80 m depth, and overall in the Mid-Atlantic
($\boldsymbol{W = e^{(\alpha + a(St) + \beta \ln(L) + \gamma ln(D)) + \epsilon}}$).



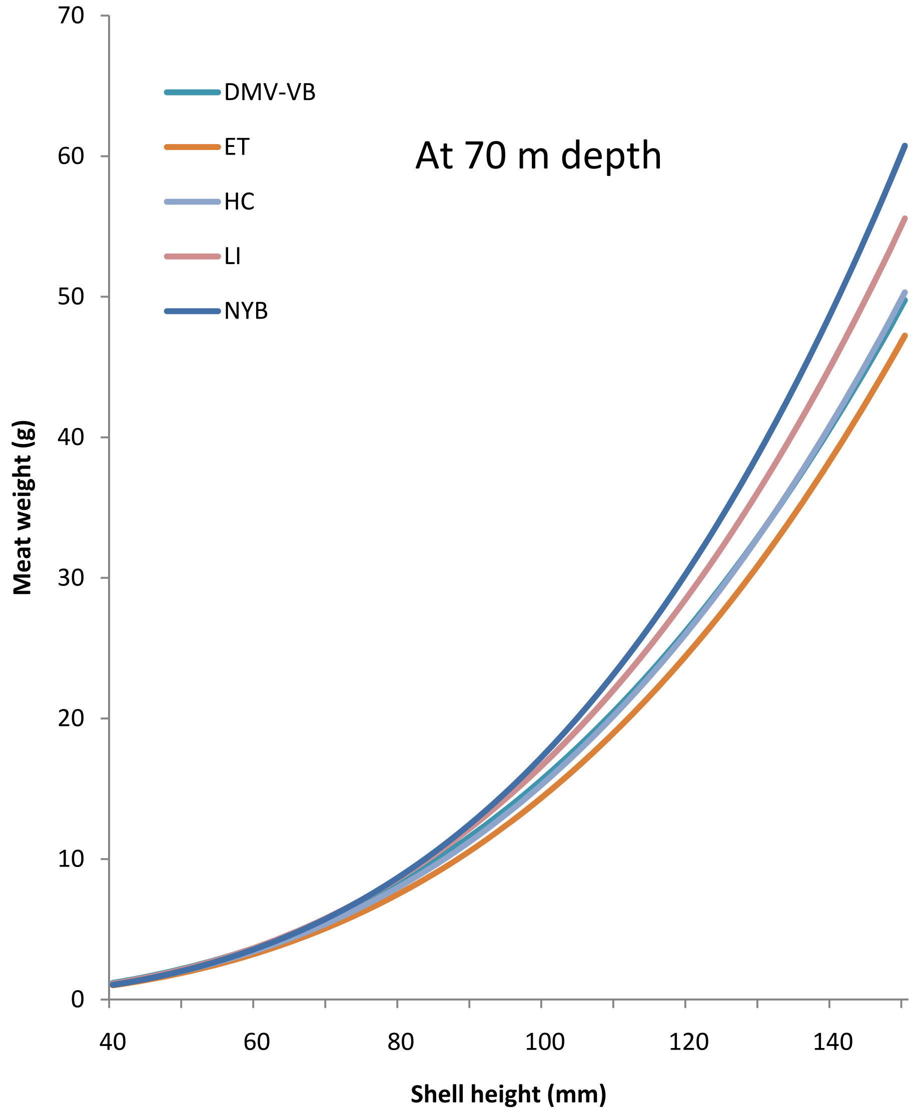
Appendix B7-Figure 7. Shell height/meat weight relationships for each survey year at 60 m depth in the Mid-Atlantic Bight ($\boldsymbol{W = e^{(\alpha + a(St) + \beta \ln(L) + \gamma ln(D)) + \epsilon}}$).



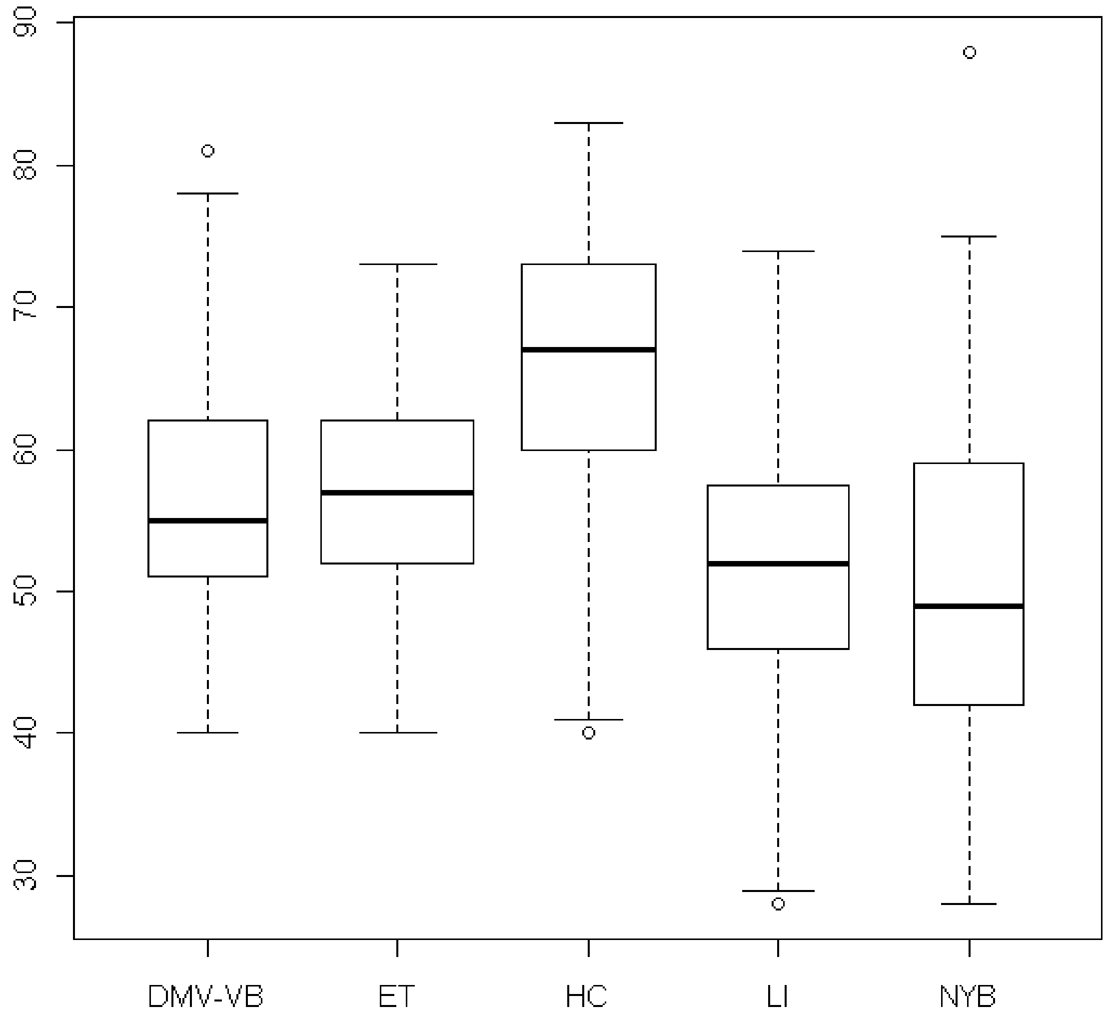
Appendix B7-Figure 8. Shell height/meat weight relationships for each subarea at 60 m depth in the Mid-Atlantic Bight ($\boldsymbol{W} = \boldsymbol{e}^{(\boldsymbol{\alpha}+\boldsymbol{a}(\boldsymbol{St})+\boldsymbol{\beta}\,\mathbf{ln}(\boldsymbol{L})+\boldsymbol{\gamma}\boldsymbol{ln}(\boldsymbol{D}))+\boldsymbol{\epsilon}}$).



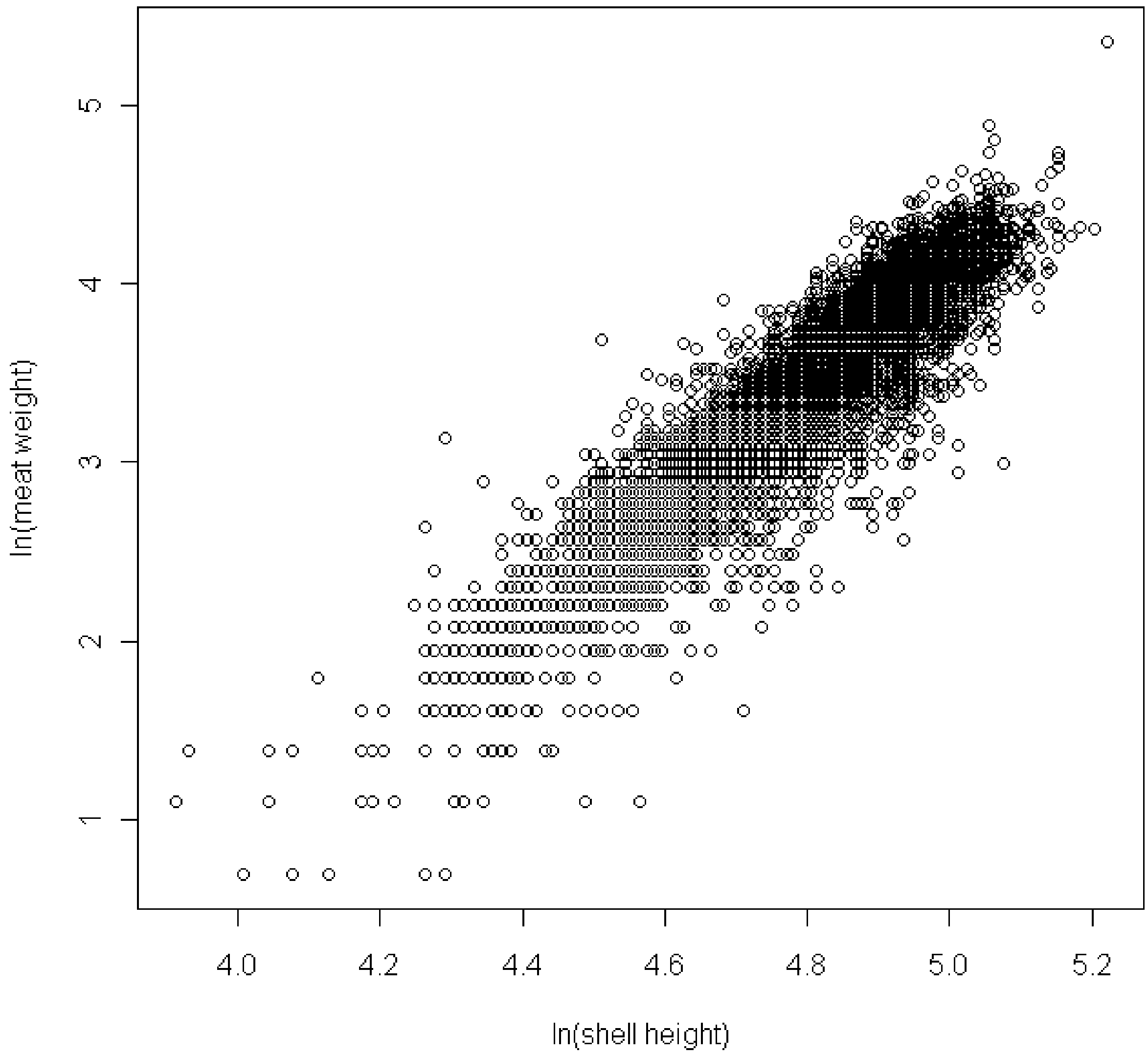
Appendix B7-Figure 9. Shell height/meat weight relationships for each subarea at 50 m depth in the Mid-Atlantic Bight ($\boldsymbol{W} = \boldsymbol{e}^{(\boldsymbol{\alpha}+\boldsymbol{a}(\boldsymbol{St})+\boldsymbol{\beta}\,\mathbf{ln}(\boldsymbol{L})+\boldsymbol{\gamma} \boldsymbol{ln}(\boldsymbol{D}))+\boldsymbol{\epsilon}}$).



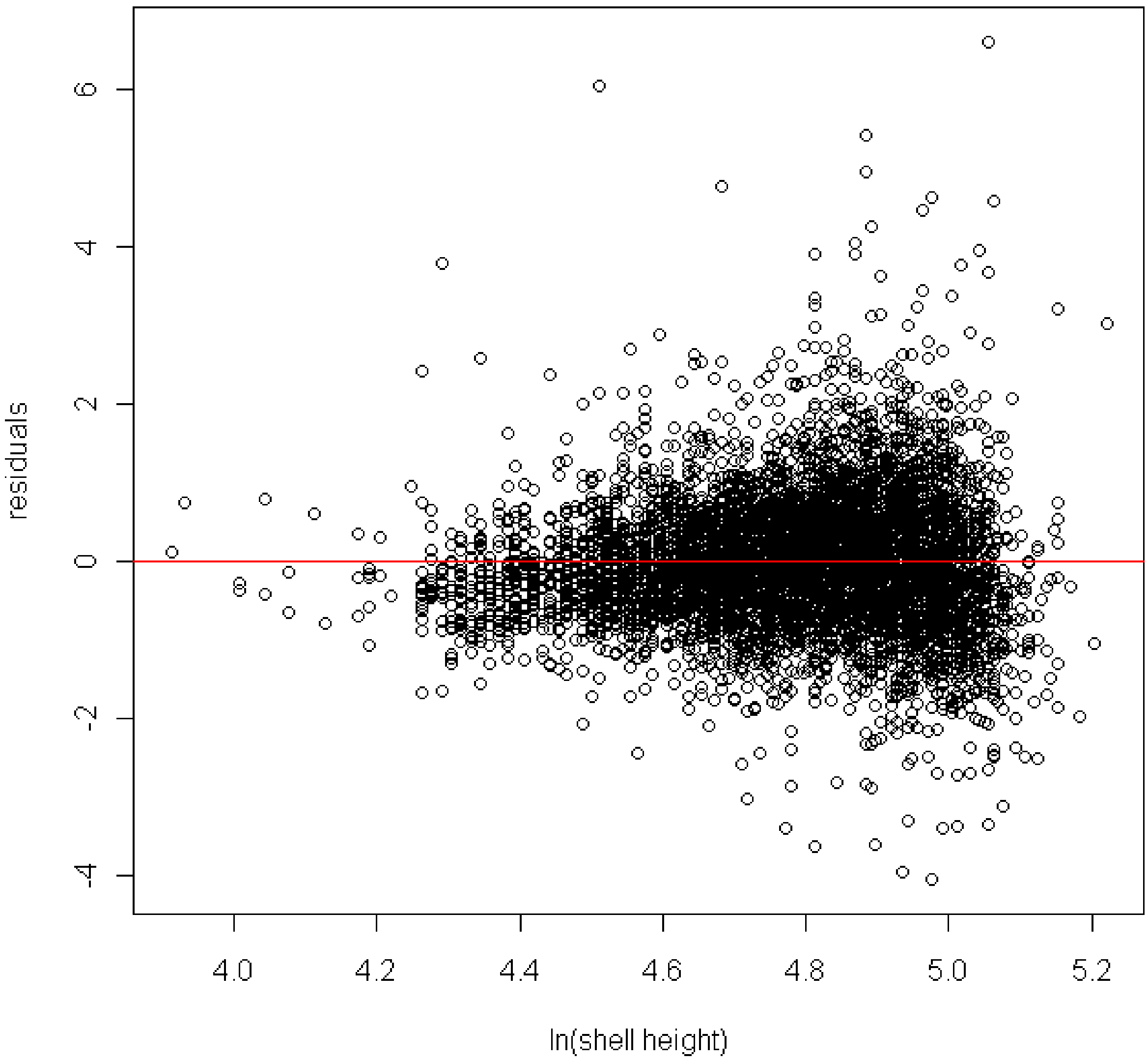
Appendix B7-Figure 10. Shell height/meat weight relationships for each subarea at 70 m depth in the Mid-Atlantic Bight ($\boldsymbol{W} = \boldsymbol{e}^{(\boldsymbol{\alpha}+\boldsymbol{a}(\boldsymbol{St})+\boldsymbol{\beta}\ln(\boldsymbol{L})+\boldsymbol{\gamma}\boldsymbol{ln}(\boldsymbol{D}))+\boldsymbol{\epsilon}}$).



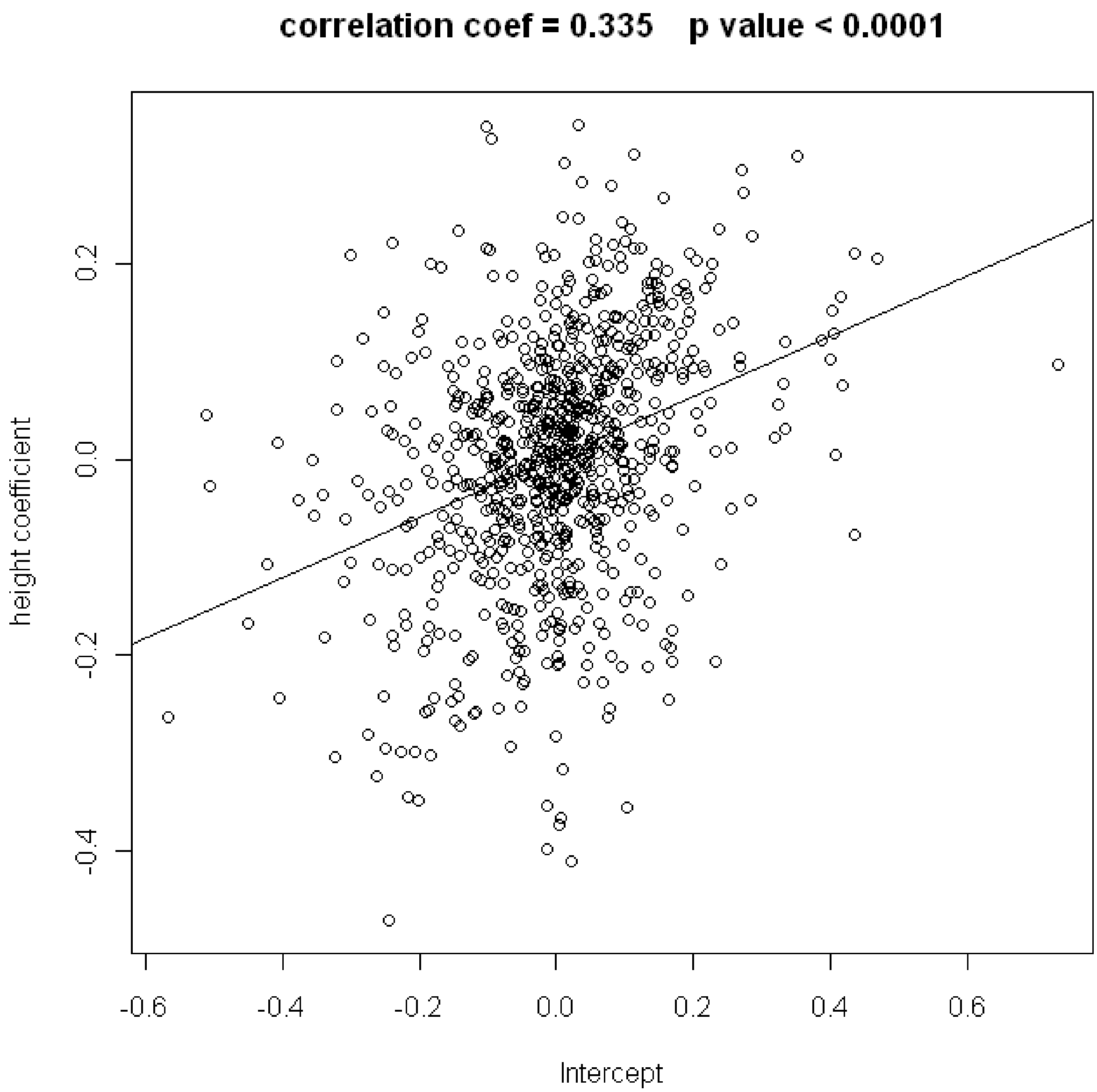
Appendix B7-Figure 11. Box plots of the depths of samples taken from each of the subareas in the Mid-Atlantic Bight.



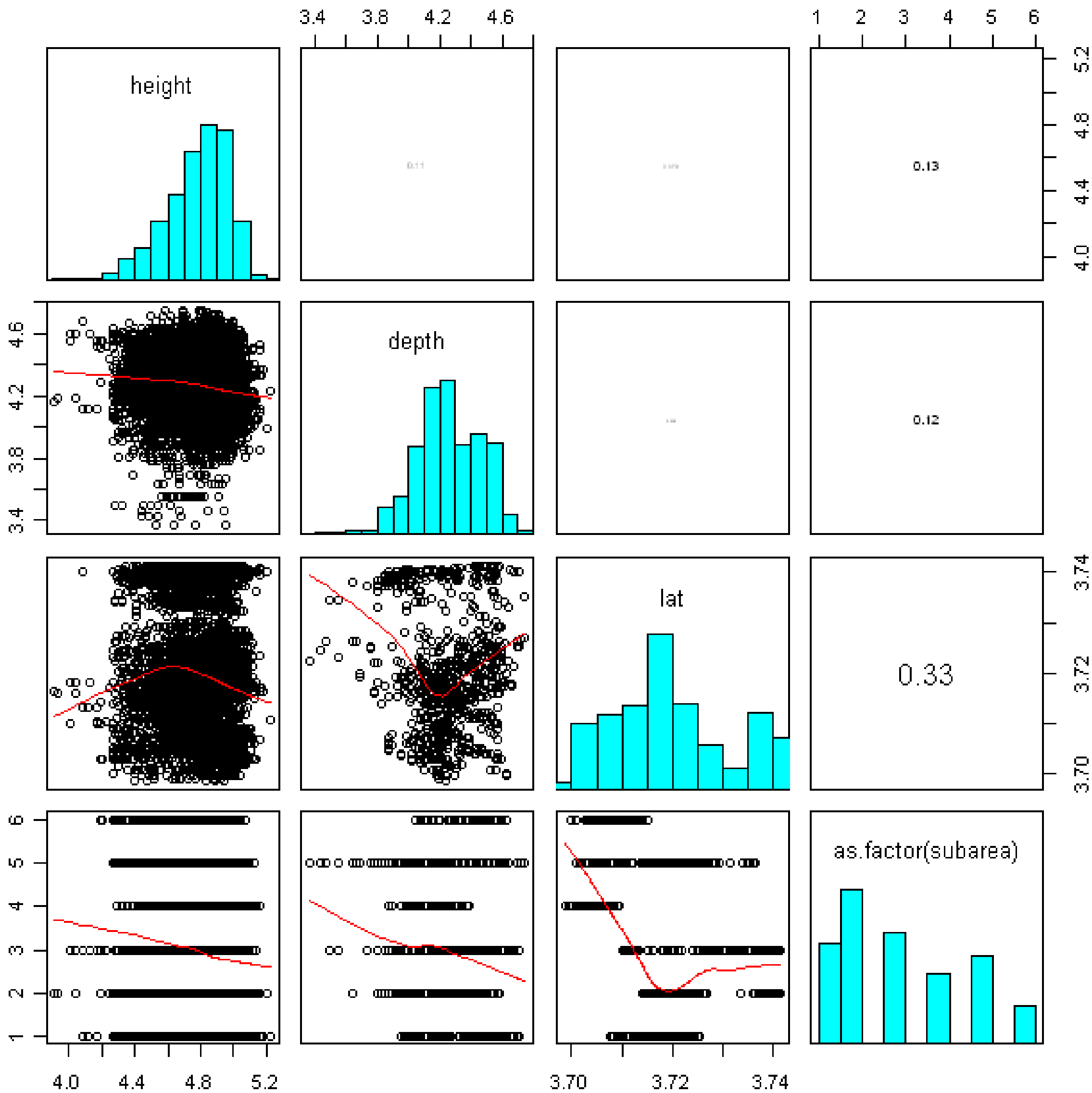
Appendix B7-Figure 12. Georges Bank shell height/meat weight data.



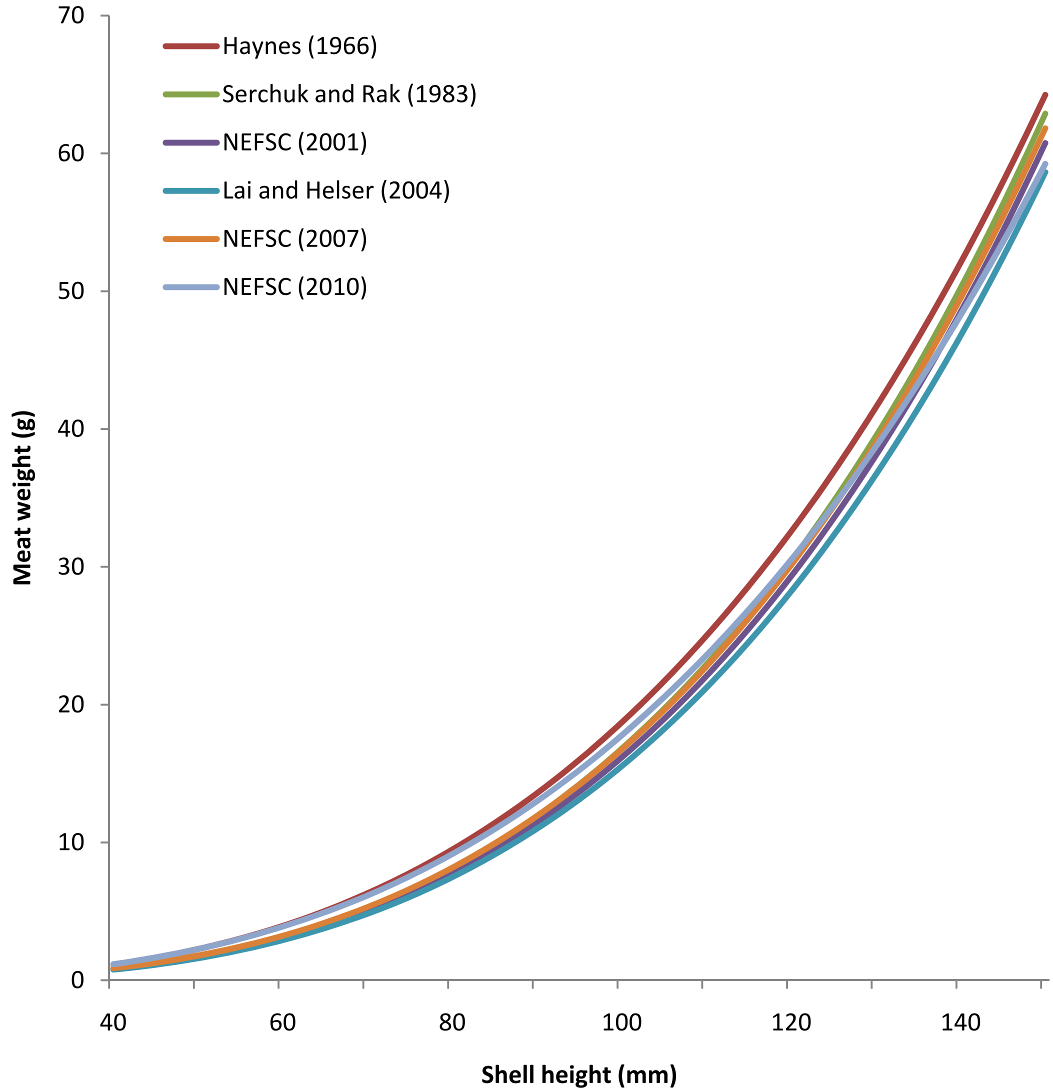
Appendix B7-Figure 13. Residual plot of Georges Bank shell height/meat weight data.



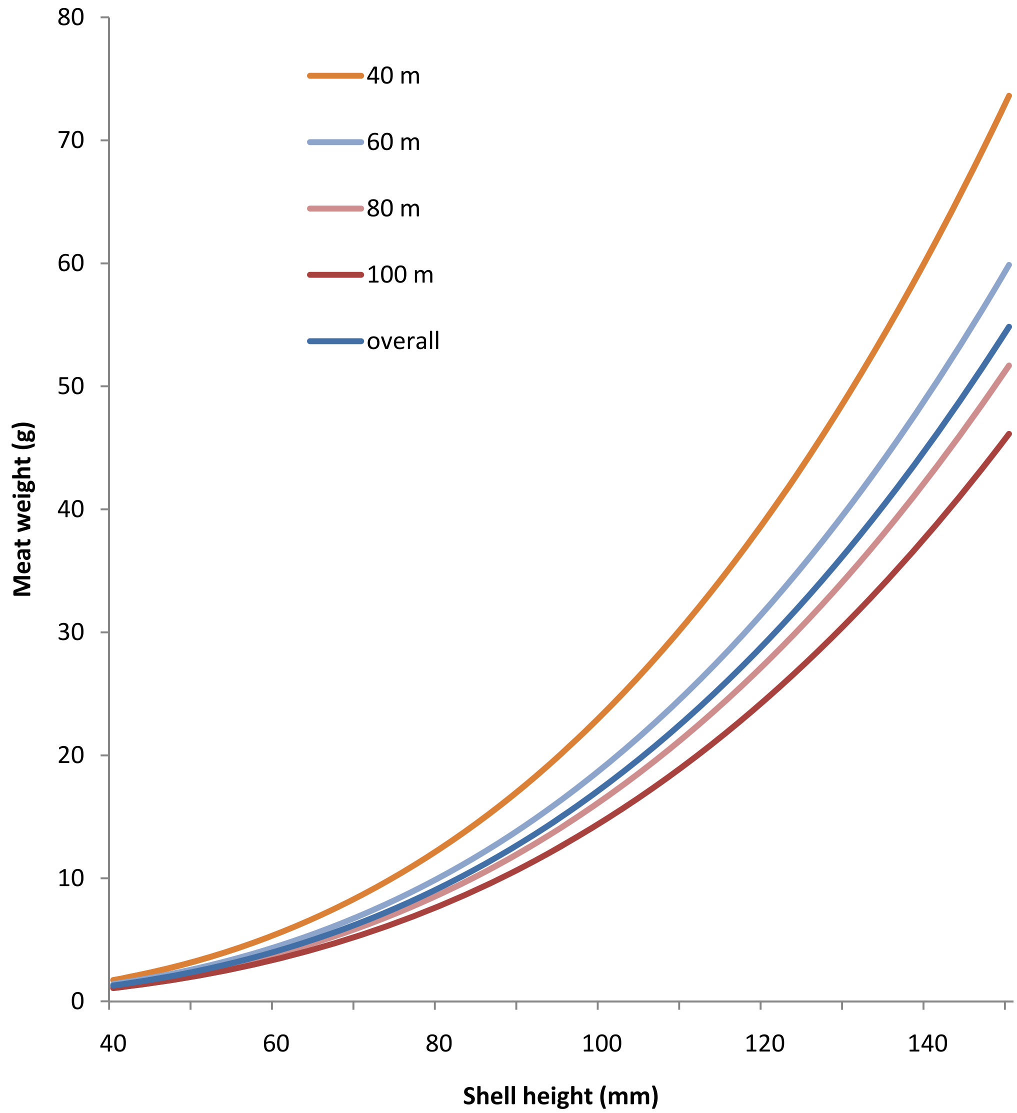
Appendix B7-Figure 14. The correlation between BLUPs (Best Linear Unbiased Predictions of random effects) from the best model (2) for Georges Bank. These are a random slope coefficient (on shell height) and a random intercept, both grouped by station (where station is a unique identifier that incorporates spatial – survey station, and temporal – year, variability).



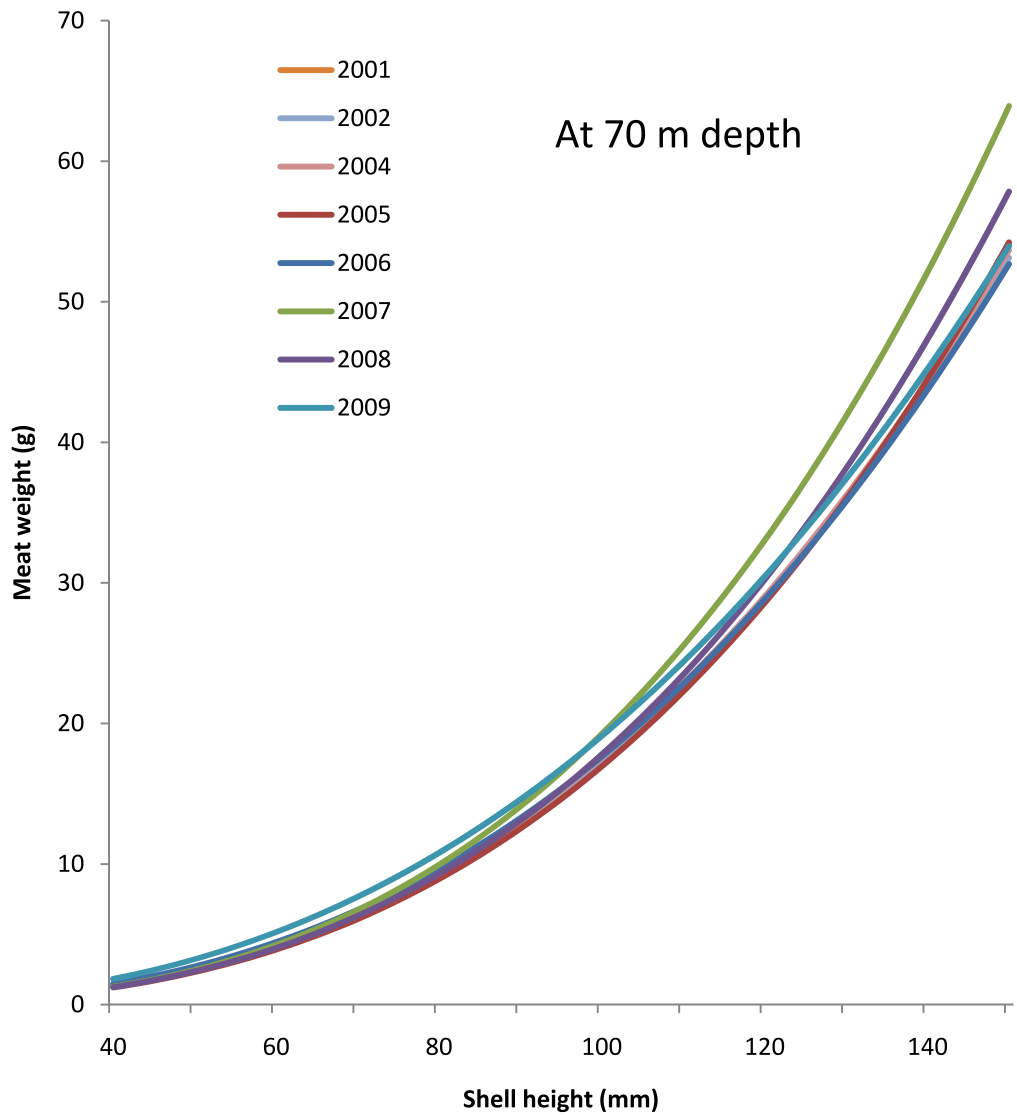
Appendix B7-Figure 15. Correlation of Fixed effects from the best model (2) for Georges Bank. The values of the correlation coefficients for each comparison are shown in the upper diagonal and the text font is scaled relative to the significance of the correlation. The main diagonal shows the frequency histogram of each effect and the scatter plots in the lower diagonal include a smooth line meant only to aid visual inspection.



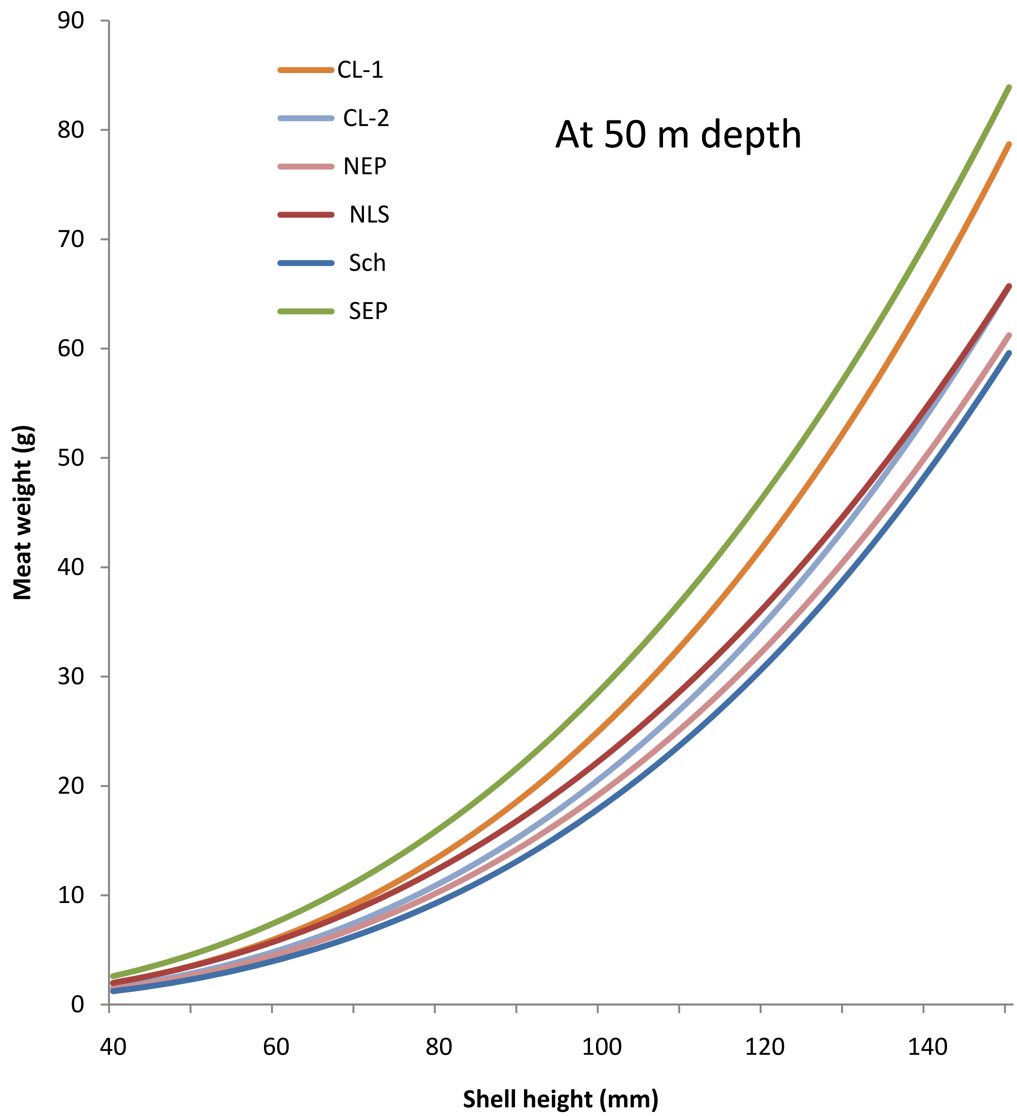
Appendix B7-Figure 16. Comparison of shell height/meat weight parameter estimates in the Georges Bank (directly comparable models only, i.e. of the form $= e^{(\alpha+a(St)+\beta)+\epsilon}$).



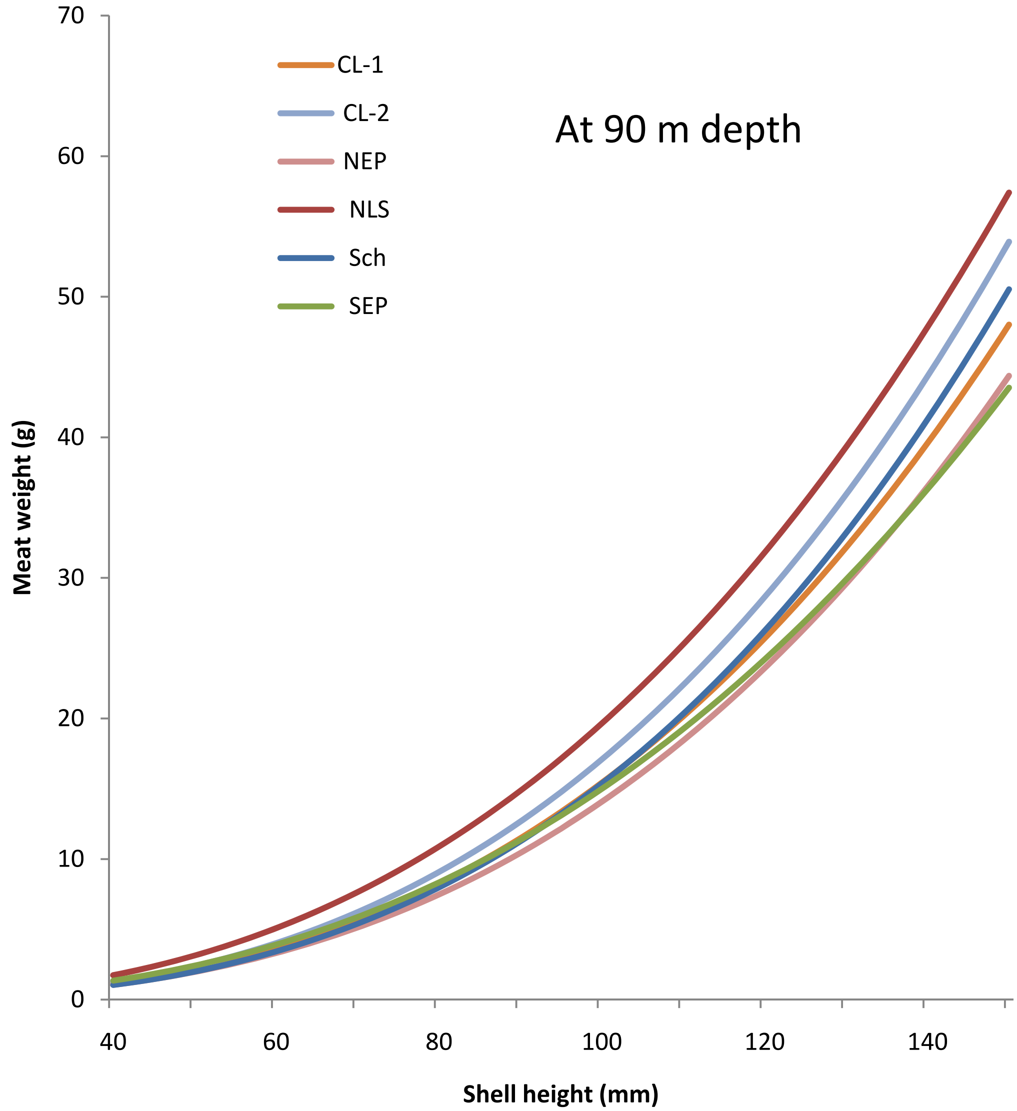
Appendix B7-Figure 17. Shell height/meat weight relationships at relationships 40, 60, 80, 100 m depth, and overall in Georges Bank ($\boldsymbol{W = e^{(\alpha + a(St) + \beta \ln(L) + \gamma ln(D) + b(L_{St})) + \epsilon}}$).



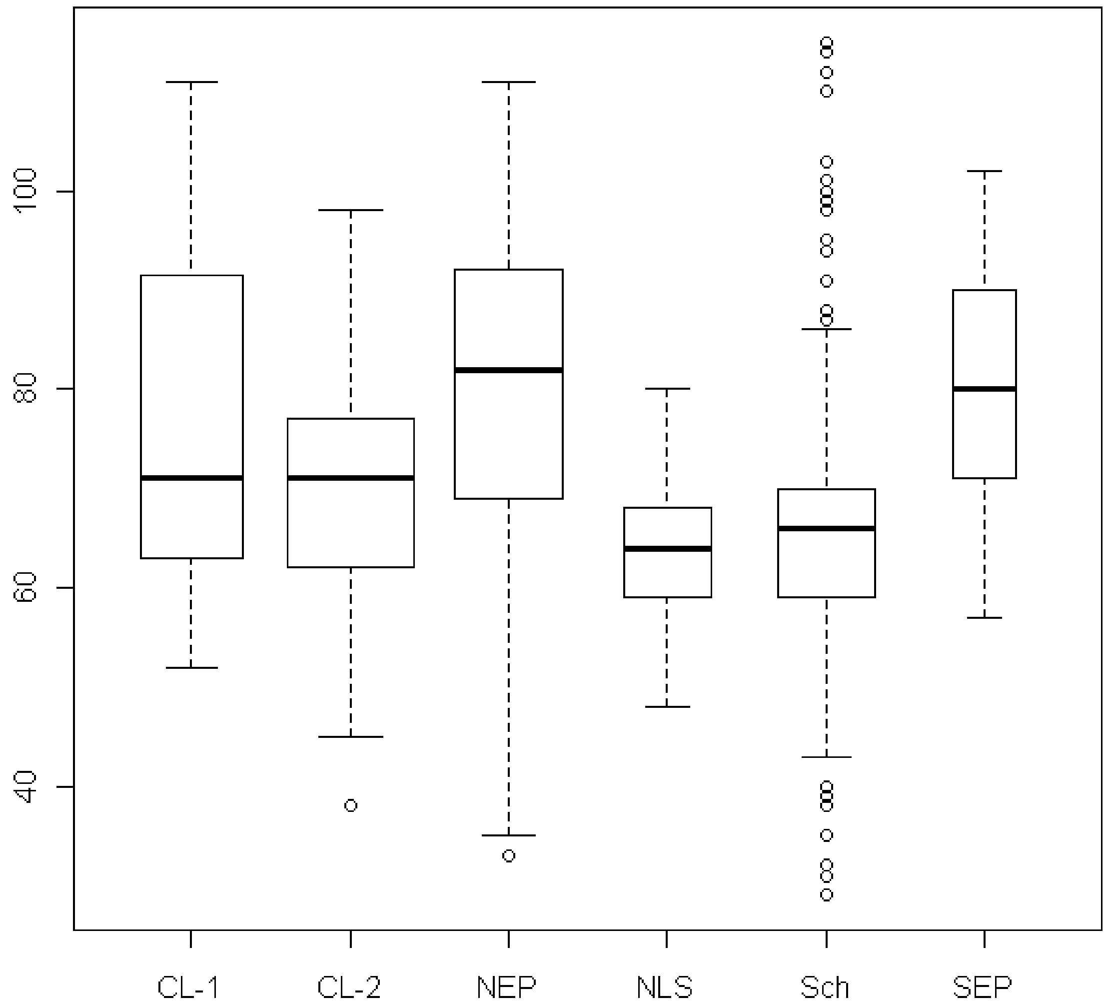
Appendix B7-Figure 18. Shell height/meat weight relationships for each survey year at 70 m depth on Georges Bank ($\boldsymbol{W} = \boldsymbol{e}^{(\boldsymbol{\alpha}+\boldsymbol{a}(\boldsymbol{St})+\boldsymbol{\beta}\ln(\boldsymbol{L})+\boldsymbol{\gamma}\boldsymbol{ln}(\boldsymbol{D})+\boldsymbol{b}(\boldsymbol{L_{St}}))+\boldsymbol{\epsilon}}$).



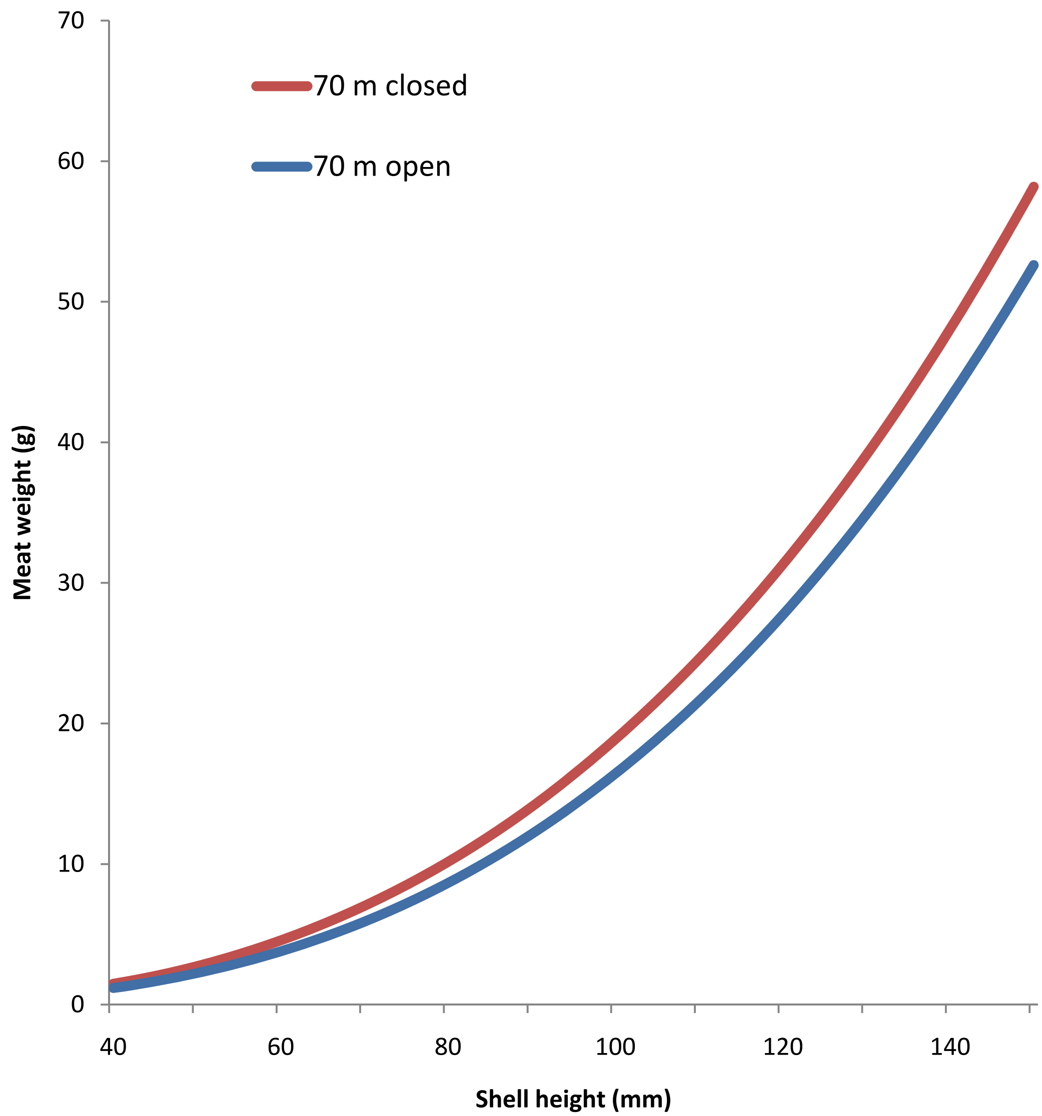
Appendix B7-Figure 19. Shell height/meat weight relationships for each survey year at 50 m depth on Georges Bank ($\boldsymbol{W} = \boldsymbol{e}^{(\alpha + a(St) + \beta \ln(L) + \gamma ln(D) + b(L_{St})) + \epsilon}$).



Appendix B7-Figure 20. Shell height/meat weight relationships for each survey year at 90 m depth on Georges Bank ($\boldsymbol{W = e^{(\alpha + a(St) + \beta \ln(L) + \gamma ln(D) + b(L_{St})) + \epsilon}}$).



Appendix B7-Figure 21. Box plots of the depths of samples taken from each of the subareas on Georges Bank.



Appendix B7--Figure 22. Shell height/meat weight relationships at relationships for open and closed to fishing areas at 60 m depth on Georges Bank ($W = e^{(\alpha + a(St) + \beta \ln(L) + \gamma ln(D) + b(L_{St})) + \epsilon}$).

Appendix B8: Seasonal patterns in commercial meat weight and meat weight anomalies.

Dan Hennen and Dvora Hart, Northeast Fisheries Science Center, Woods Hole, MA.

This appendix describes updated estimates of seasonal patterns in mean commercial meat weights and updated annual commercial meat weight anomalies. The anomalies are used in the CASA model (Appendix 11) in calculating predicted catch weight to account for differences in shell-height meat weight relationships between the NEFSC scallop survey and commercial fishery. Relationships from the NEFSC scallop survey are used to calculate mid-year biomass for the population. Anomalies for the commercial fishery are calculated on an annual basis to account for overall and seasonal differences in survey and commercial meat weight, and changes over time in the seasonal distribution of catches.

Methods

The NMFS Observer program provided meat weight estimates from commercial catches that occurred throughout the year. These meat weights are for sea scallops in samples that are shucked by fishermen after the observer measures shell height. Meats from the observer program are not weighed individually. They are packed into a graduated cylinder and a volume for a sample (typically ~100 scallops) is recorded. The meat weight for a sample is calculated assuming a density estimate of 1.05 g/ml^3 (Caddy and Radley-Walters 1972; Smolowitz et al. 1989). Shell height data from the observer program for individual scallops are binned by 5 mm increments.

Predicted meat weights for the Mid-Atlantic Bight (MAB) and Georges Bank (GBK) were based on the models

$$W = e^{\left(\alpha+\beta\ln(H)+\gamma ln(D)+\rho(\ln(L)*\ln(D))\right)+\epsilon} \qquad \text{(MAB)} \qquad (1)$$

$$W = e^{(\alpha+\beta\ln(H)+\gamma ln(D)+\rho ln(L))+\epsilon} \qquad \text{(GBK)} \qquad (2)$$

where W is meat weight (g), H is shell height in mm, D is depth in m, and L is latitude measured in decimal degrees. This model was fit using NMFS scallop survey data from 2001 – 2008 (Appendix B7). As described in NEFSC (2007), the surveys for scallops occur in the summer when meat weights are typically high. The estimated coefficients from (1) and (2) were applied to the shell heights and depths recorded from observer samples from 2001 – 2009. Observer data for 2006 is incomplete and was not used in this analysis. Monthly anomalies were computed using median predicted meat weights and median meat weights derived from observer data:

$$\frac{(pred. - obs.)}{pred.} \tag{3}$$

Median meat weights were used instead of mean meat weights to reduce the influence of outliers in the data. In general, the observed meat weights (from observed volumes) should be less than the survey-based, predicted meat weights because the commercially shucked scallops leave some meat on the shell, and because the surveys occur in mid to late summer, a time of typically high meat weight. For both the Mid-Atlantic and Georges Bank, however, there were months of the year where the observed scallop meats were heavier than the predicted meats. In the Mid-Atlantic, peak meat weight occurred in April through August (Figure 1), while on Georges Bank peak meat weight occurred in June (Figure 2).

There are differences in the month in which peak meat weight occurs over the years of the study (Figures 3 and 4). Peak meat weight appears to have occurred earlier during recent years, though the time series is too short and there are too few observations to provide precise estimates of seasonal patterns on an annual basis. The typical seasonal pattern is therefore used in calculating anomalies for all years.

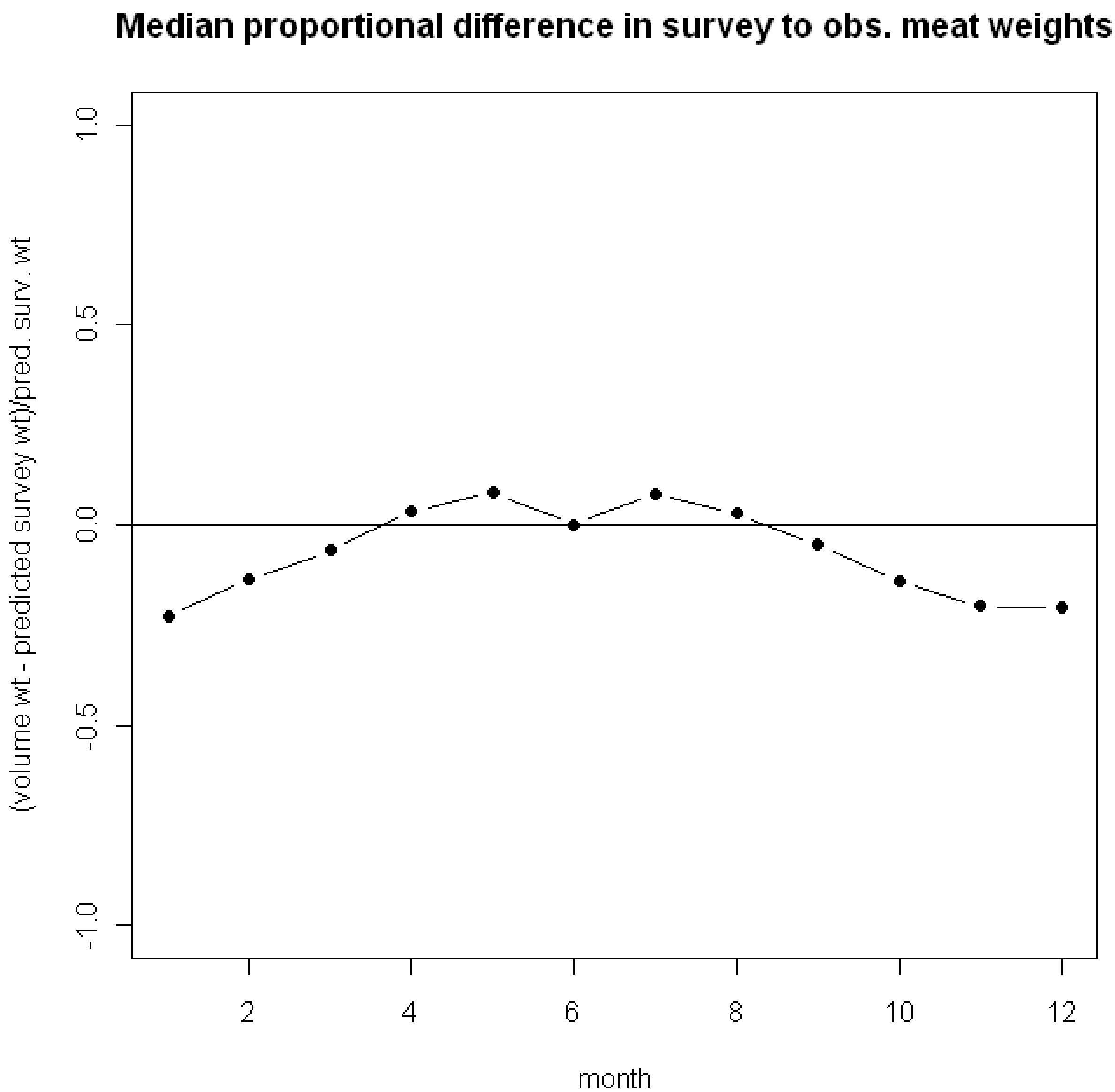
Median meat weight anomalies for 2003-2008 were smoothed by a second order polynomial loess function with a span of 0.25 (months). This short smoothing span provided a modest smooth that allowed the data to strongly influence the model fit (Figures 5 and 6). The smooth was applied to a duplicated annual cycle (i.e. 24 months were fit, using identical data in each 12 month period) and the middle 12 months were selected and reordered so that January was the first month in the resulting model fit. This manipulation guaranteed that December and January produced linking estimates. The smoothed monthly anomalies were then weighted by the landings in each month in each year for which we have landings data (1975 – 2008) and annual median values were calculated.

Updated annual meat weight anomalies differ from those in the last assessment (Figures 7 and 8). The updated anomalies are generally higher in the MAB (~7% higher on average) and lower in the GBK (~8% lower on average). In MAB the differences are due to new observer data which reflect an increase in meat weights during 2007-2008 (Figure 9). In GBK, 2007 and 2008 had relatively heavy survey meat weights (Figure 11). These two years are 40% of the years considered in this analysis. Therefore the meat weight trends in recent survey years are influential.

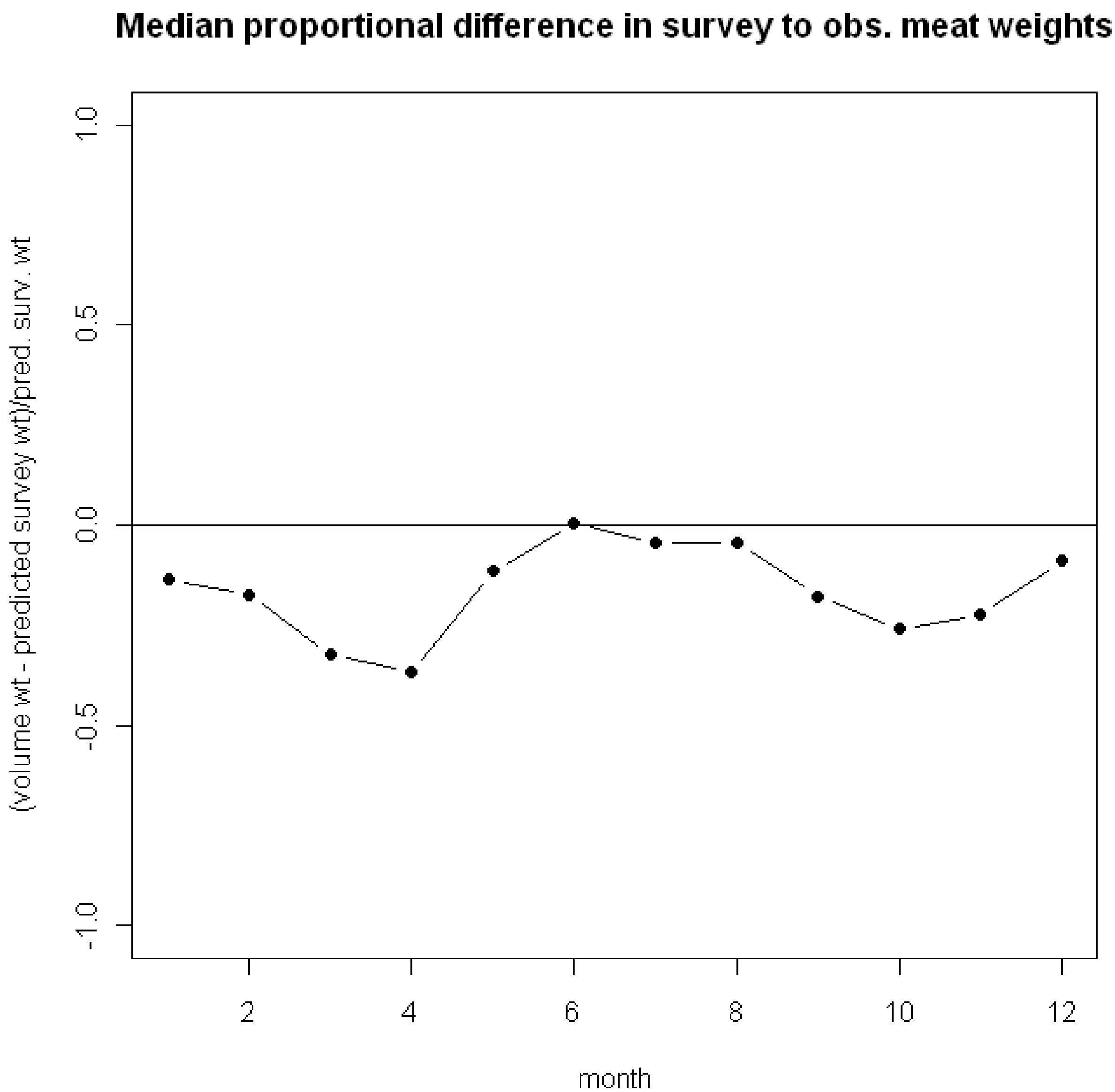
Literature Cited

Caddy, J.F. and C. Radley-Walters. 1972. Estimating count per pound of scallop meats by volumetric measurement. Fish. Res. Brd. Can. Man. Rep. Ser. 1202.

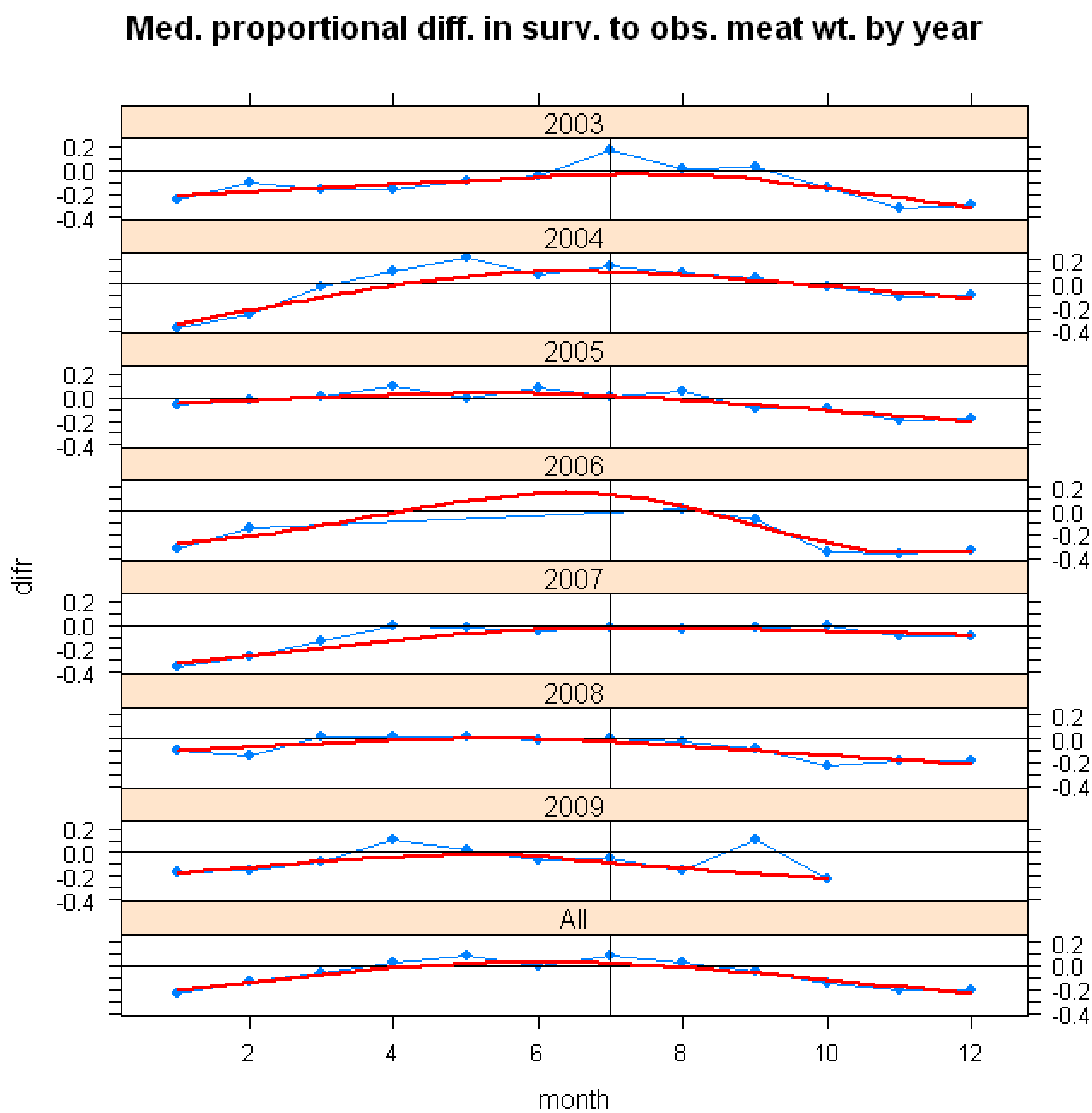
Smolowitz, R.J., F.M. Serchuk and R.J. Reidman. 1989. The use of a volumetric measure for determining sea scallop meat count. NOAA Tech. Mem. F/NER-1.



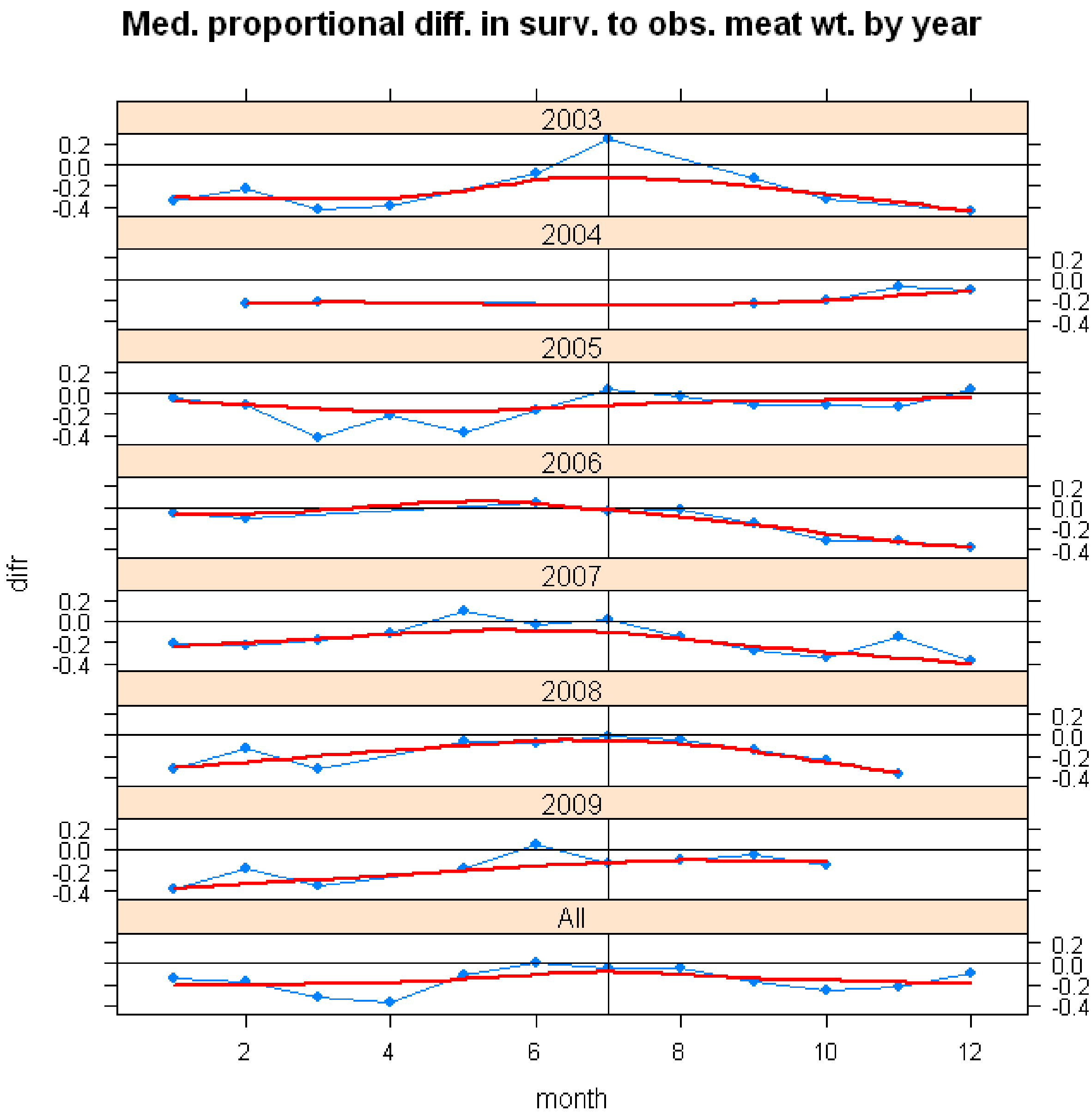
Appendix B8-Figure 1. Meat weight anomalies by month for the Mid-Atlantic Bight.



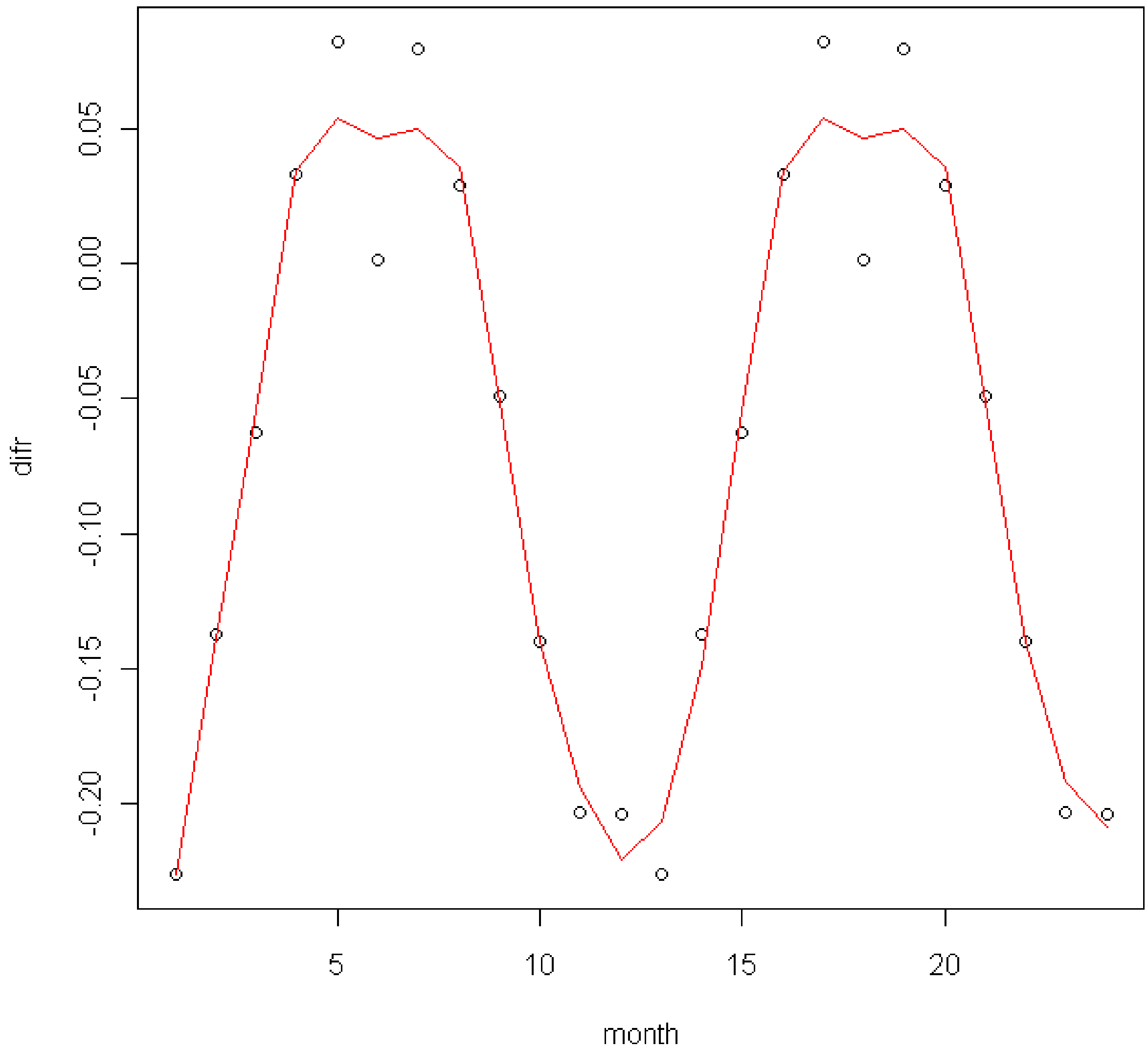
Appendix B8-Figure 2. Meat weight anomalies by month for Georges Bank.



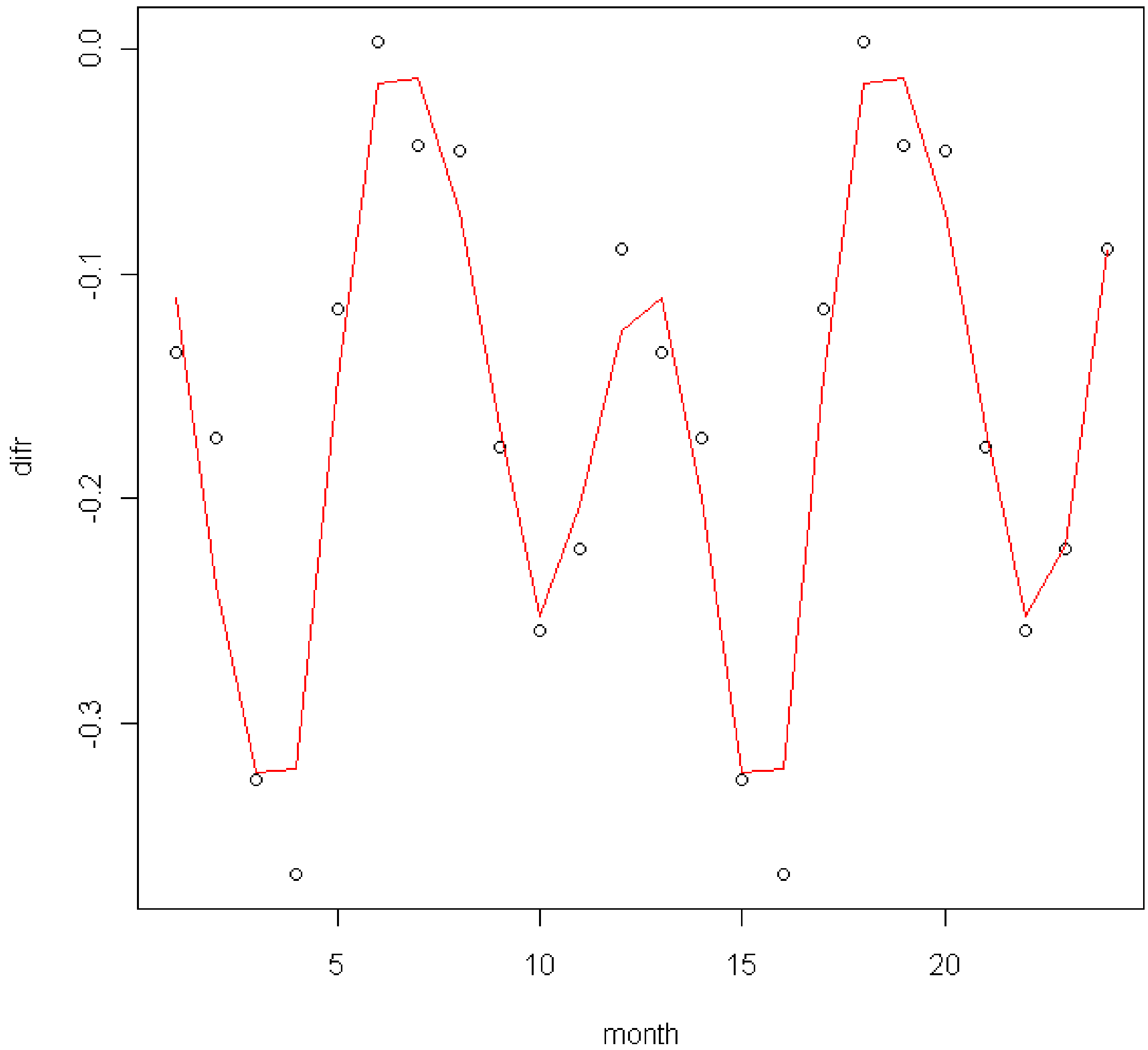
Appendix B8-Figure 3. Observer predicted meat weights (based on volume) compared to meat weights predicted by a model based survey data, by month, year, and overall, from the Mid-Atlantic. The red line is a loess regression and is used only to illustrate seasonal trends.



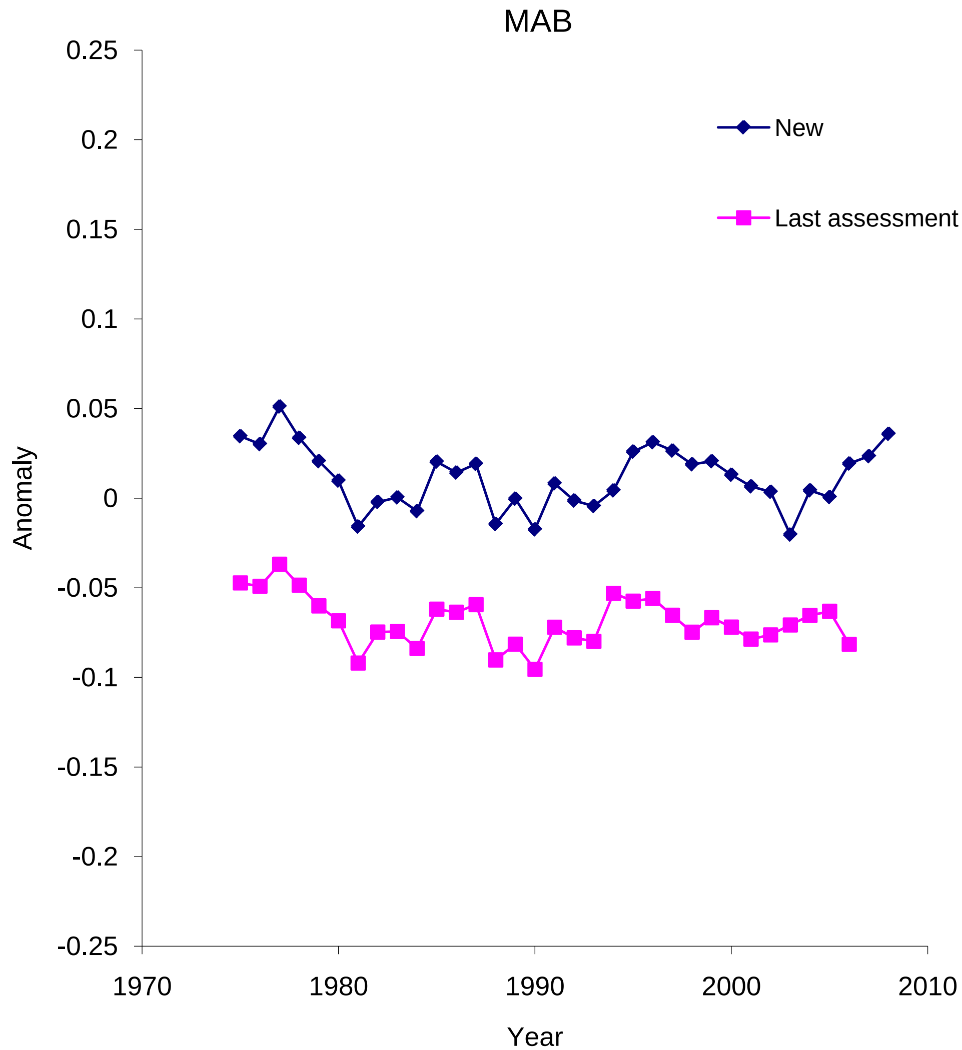
Appendix B8-Figure 4. Observer predicted meat weights (based on volume) compared to meat weights predicted by a model based on survey data, by month, in each year, and overall, from Georges Bank. The red line is a loess regression and is used only to illustrate seasonal trends.



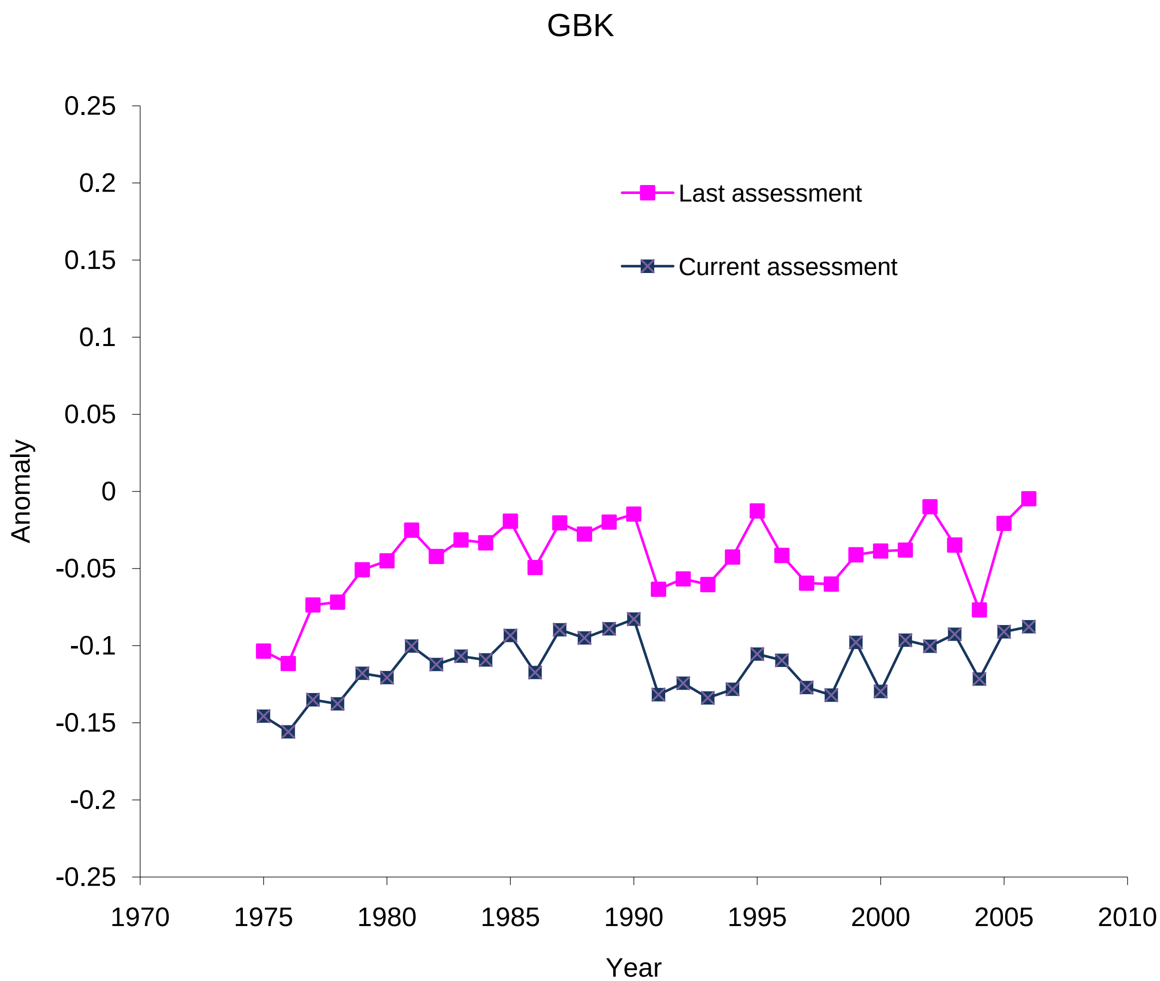
Appendix B8-Figure 5. Smoothed meat weight anomalies by month in the MAB.



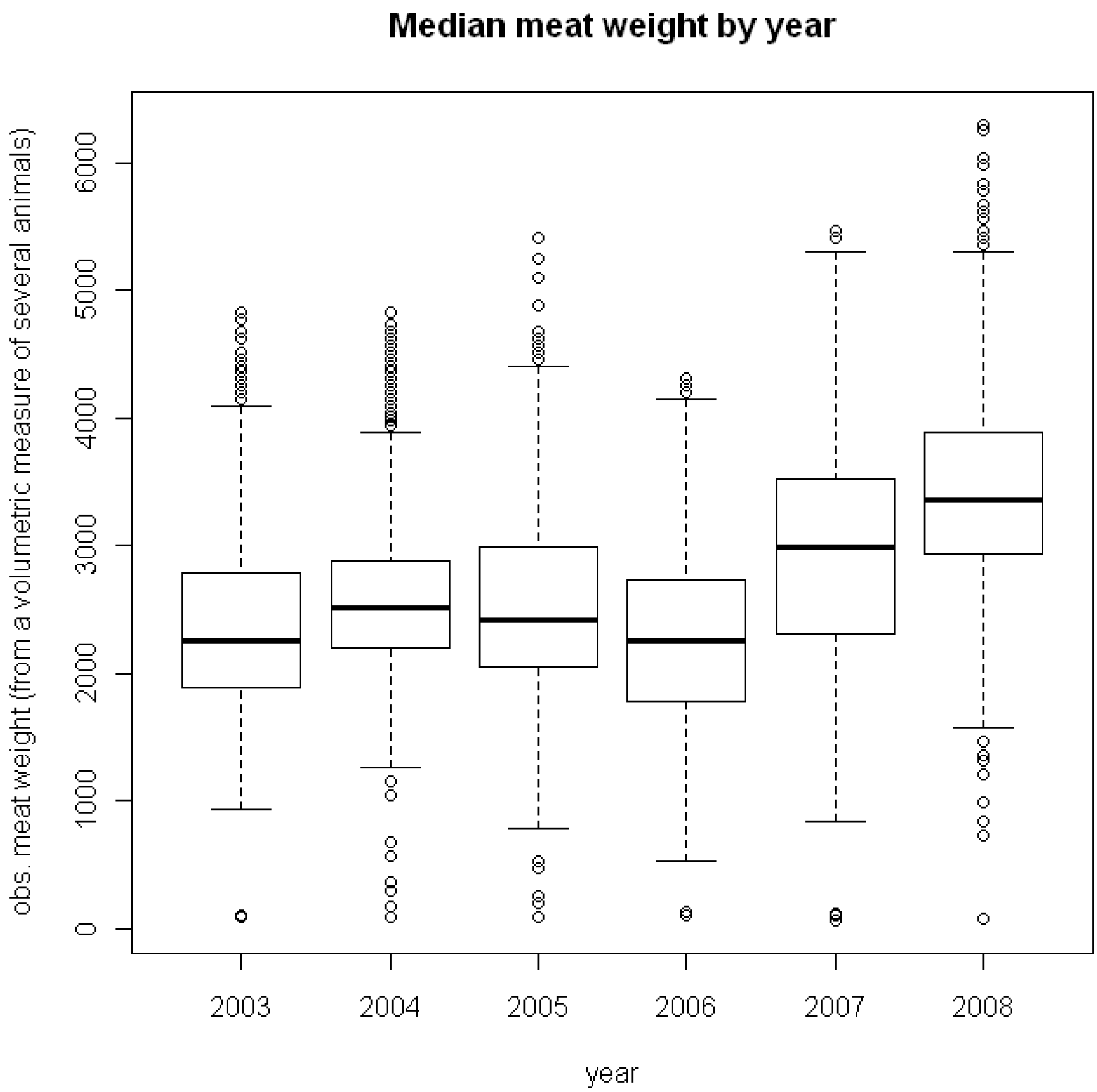
Appendix B8-Figure 6. Smoothed meat weight anomalies by month in the MAB.



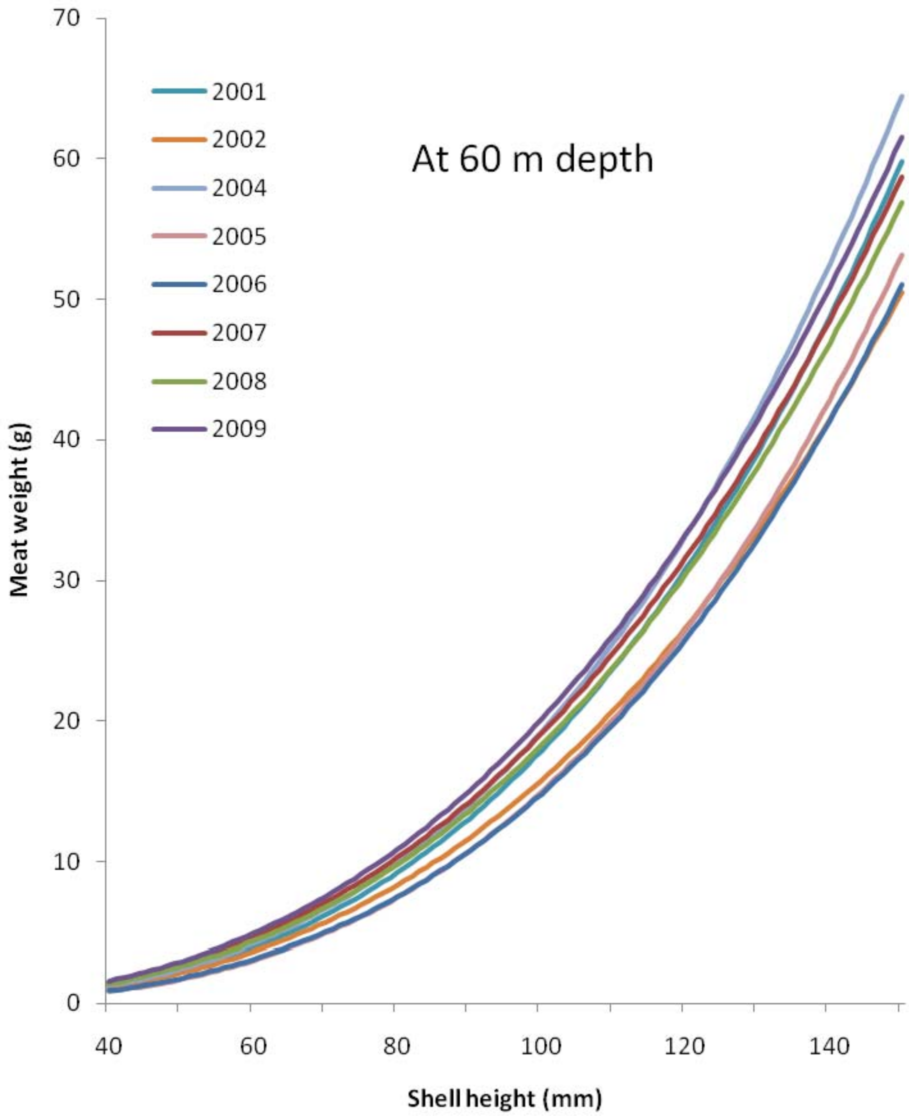
Appendix B8-Figure **7**. A comparison between the meat weight anomaly (smoothed and weighted by landings in each month) by year, as calculated in the last assessment and the current meat weight anomaly in the MAB.



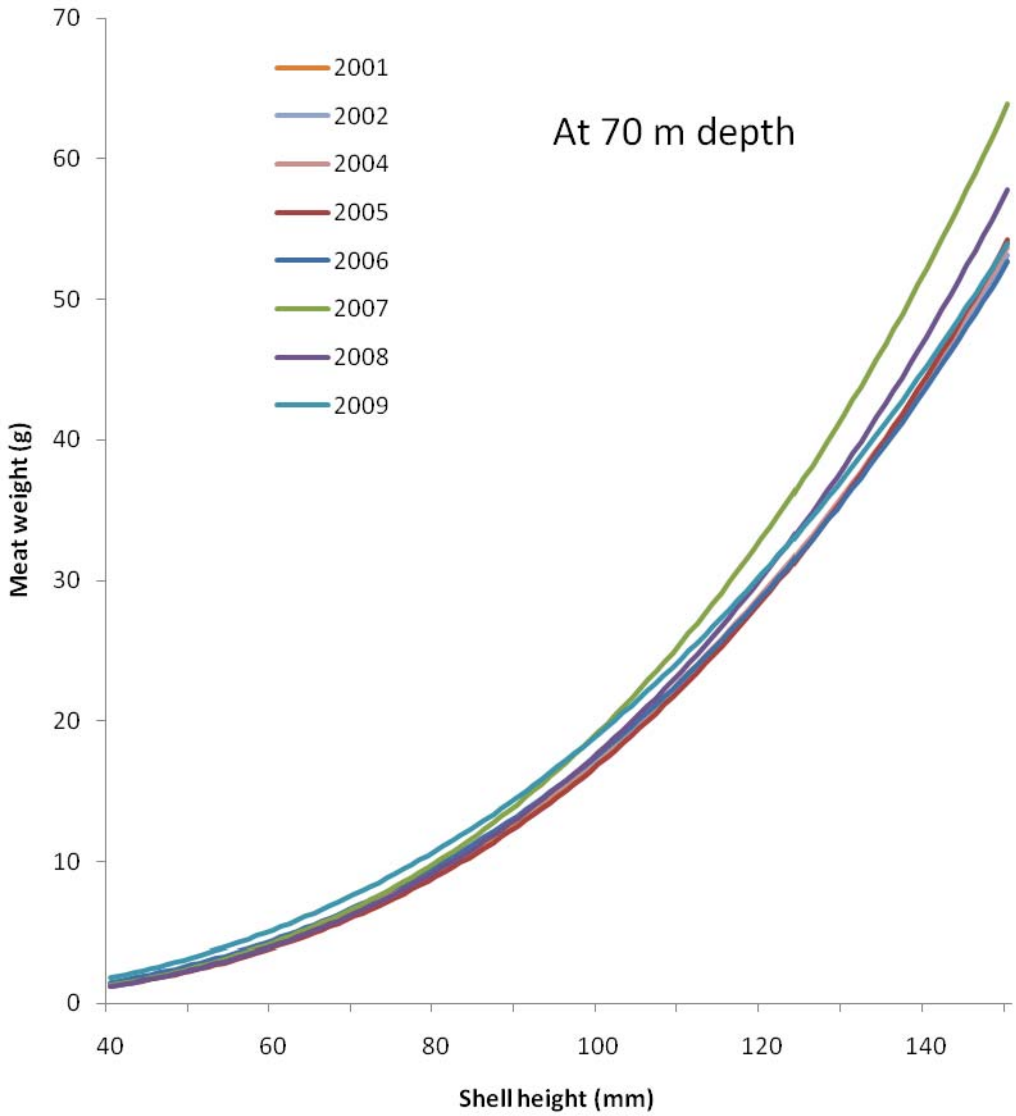
Appendix B8-Figure 8. A comparison between the meat weight anomaly (smoothed and weighted by landings in each month) by year, as calculated in the last assessment and the current meat weight anomaly in the GBK.



Appendix B8-Figure 9. The observed meat weight in the commercial catch by year. Observed meat weights are based on a simple density conversion of the volume of approximately 100 commercially shucked meats.



Appendix B8-Figure 10. Shell height/meat weight relationships for each survey year at 60 m depth in the Mid-Atlantic Bight ($W = e^{(\alpha + a(St) + \beta\, ln(L) + \gamma ln(D)) + \epsilon}$).



Appendix B8-Figure 11. Shell height-meat weight relationships by survey year at 70 m depth on Georges Bank ($W=e^{(\alpha+a(St)+\beta\ln(L)+\gamma\ln(D)+b(L_{St}))+\epsilon}$).

Appendix B9: Summary of HabCam survey results for sea scallops and yellowtail flounder in the Nantucket Lightship Closed Area during 2009

The HabCam Group
Scott M. Gallager[1, 2], Amber D. York[1, 2]
Richard Taylor[2, 3], Norman Vine[2]
Jonathan Howland[1], Steve Lerner[1]
Karen Bolles[3, 4]

(1) Woods Hole Oceanographic Institution
(2) Advanced Habitat Imaging Consortium
(3) Arnie's Fisheries, New Bedford, MA
(4) Ocean Explorium, New Bedford, MA

Conclusions

HabCam is a cabled optical and acoustic imaging system that is "flown" from a ship traveling at 5 kn at an altitude of 1 to 3 meters off the bottom while collecting high resolution still images at a rate of six images per second. Imaging rate provides ~50% overlap to allow for construction of image mosaics of the seafloor. A track approximately 100 nautical miles in length and 259,200 m^2 in area is imaged each 24 hour day while at sea. When operating continuously, HabCam samples nearly 2.5 times the area covered by a survey dredge.

Manual classification of the images provides the following information: 1) counts and measurements on sea scallops and groundfish (i.e. cod, haddock, flounders), epibenthic megafauna and many benthic infaunal species; 2) characterization of substrate; 3) observations on animal behaviors, inter- and intra-species interactions, biodiversity and community structure; 4) the ability to assess and monitor invasive species; 5) the tools to characterize oceanic properties (salinity, temperature, nutrients); and 6) the means to "map" the location of lost fishing gear (e.g. trawl and gillnets, lobster pots, and other miscellaneous fishing gear and parts). Automated methods for target classification are currently under development and will provide tools to reprocess archived results of image surveys as new technologies are developed.

Here we report on use of the HabCam camera system to: (1) conduct sea scallop surveys in the Nantucket Lightship Closed Area (NLSCA) as part of an effort to compare sampling technologies, (2) conduct dredge calibration with NOAA/NMFS vessels, and (3) conduct an analysis of inherent errors in camera calibration, scallop abundance estimates, and shell height measurements.

The objectives of the 2009 NLSCA survey were to estimate scallop abundance, shell height frequency distribution and biomass, and to estimate the distribution and abundance of yellowtail flounder in relation to substrate. A survey track line was designed as a modified spiral with track spacing from 2.6 to 1.3 nm. Total track line length was 348 nm with 1,235,251 images collected. Every 10th image was processed for a total of 123,000 images resulting in a total area covered of 0.187 nm^2 or 0.57% of the NLSCA. The density of scallops along survey tracks ranged between 0 and 23 scallops/m^2 with dense aggregations occurring at patch scales of about 400 and 900m. The raw values for scallop abundance on a per image basis were interpolated across the closed area using ordinary kriging. Total number of scallops in the NLSCA was 197,545,580. The overall mean for the closed area was 0.187 scallops/m^2 with a CV of 0.04.

Variance between cells ranged between 0.5, where the sampling density was highest, to over 0.7 where the sampling density was lower. A simple, alternative method to kriging for calculating total scallop counts is to multiply the mean abundance by the area. Our results using this approach is 0.187 scallops/m^2 x 1,142,280,000 m^2 = 213,606,360 scallops with a CV of 0.034, which is similar to the value estimated by kriging. Mean biomass per scallop was 32.9 g. Scallop meat biomass estimated by kriging in the closed area was 6,782 MT.

The HabCam system may be useful for mobile demersal fish, such as yellowtail flounder, in addition to sessile organisms. In the NLSCA, 124 observations of yellowtail flounder were made with the densest concentration in the central region. This region was also characterized by being mostly sand with patches of gravel. The most abundant aggregations of yellowtail were observed in the Southeast Part of CLAII and on the Canadian side with densities exceeding 0.14 fish/m^2.

Since 2007, joint tows between the NMFS annual dredge survey and HabCam have been designed to compare scallop abundances and size estimates from the standard federal dredge survey with those derived from HabCam. HabCam estimates of scallop abundance were consistently greater than those for the dredge. Mean shell height measurements were similar between dredge and HabCam but the tails of the frequency distribution for HabCam are higher than those for the dredge indicating some inherent measurement error.

In June 2009, in addition to shadowing the Sharp with the Kathy Marie during Legs 1 and 2 of the annual scallop survey, HabCam was towed from the A-frame of the Sharp as part of routine dredge operations on Leg 3. Regression slopes between dredge and HabCam scallop abundances were 0.34 for Georges Bank stations and 0.46 for Mid Atlantic Bight stations. When the data are broken out by substrate regardless of region, the regression slope for sand was 0.35. For sand plus other substrate types such as shell hash the slope was 0.40 and on gravel it was 0.35. These results are a simple measure of the sampling efficiency of the dredge relative to HabCam but are biased low: see Appendix X for an unbiased approach. Results (dredge sampling efficiencies for sea scallops ~ 0.3 to 0.45) are similar to results from other studies. Moreover, they illustrate the potential for use of HabCam in directly estimating the sampling efficiency of other types of survey and fishing gear.

Errors associated with camera calibration and manual measurement of scallop shells on the computer screen were assessed. Camera resolution depends on altitude off the bottom and ranges between 0.37 – 0.89 mm/pixel. Following camera calibration intrinsic pixel error was +/- 1.59 pixels resulting in a real-world error of 0.58 – 1.41 mm. Extrinsic errors associated with geometric projection of the image plane on the seafloor, taking into account of vehicle roll, pitch, and changes in altitude, produces real-world errors of 1.11 – 1.78 mm under optimal water quality conditions in a test tank.

To estimate the level of error associated with manual screen measurement of scallops both within and between a given technician, we assigned four identical 4.2 nm long image transects containing 4,432 images from Western Great South Channel to four technicians. In most cases, the mean shell height within technician measurements were either accurate to the same number of pixels or within one pixel suggesting that within technician variability was extremely low. However, between technician variability was greater with a overall error ranging +/- 4 pixels, which represents a real-world error of 3.0 to 7.1 mm. Therefore, measurement errors of scallop shell height are dominated by human extraction of data from the images.

Introduction

There is a great need in fisheries science to develop and utilize new tools and technologies that could help improve the assessment and management of our national marine resources. Coupled with this is a major change in approach from single species to ecosystem based management. The HabCam system was developed to move toward these goals.

HabCam is a seafloor imaging camera system mounted in a ten foot steel frame, and towed at about five knots 1 to 3 m off the ocean floor. HabCam is normally towed behind the F/V Kathy Marie, a New Bedford sea scallop vessel and can operate over the range of the continental shelf, 20 to 250 m depth. The HabCam Group consists of independent researchers, Woods Hole Oceanographic Institution engineers and scientists, and fishermen. HabCam was initially designed and constructed with funding from the Northeast Consortium with major improvements made with funding from the Scallop Research Set Aside Program. The initial goal of the HabCam project was to help improve sea scallop stock assessments by increasing the accuracy of scallop biomass estimates. Additional funding from the NOAA Integrated Ocean Observing Systems (IOOS) Program to support the Northeast Bentho-pelagic Observatory (NEBO) has greatly expanded the range of uses of the HabCam instrument. For example, small study areas have been revisited seasonally providing the baseline of an ecological time series.

Attributes of the HabCam system include: 1) acquisition of optical and acoustic imagery which can be viewed in "real time", 2) the ability to count and measure scallops and groundfish (i.e. cod, haddock, flounders), 3) measurement of biodiversity and community structure, 4) the means to "map" where there are lost fishing gears (e.g. trawl and gillnets, lobster pots, and other miscellaneous fishing gear and parts), 5) characterization of substrate 6) measurement of oceanic properties (salinity, temperature, nutrients) 7) availability of data and data products online, and 8) relatively inexpensive operating costs. The HabCam system also has the ability to observe animal behaviors including inter- and intraspecies interactions as well as assess and monitor invasive species such as *Didemnum vexillum* and other epibenthic megafauna, and benthic infaunal species.

A historic record of images will be beneficial to understanding patterns, particularly in the implementation of ecosystem management schema. Further, because of direct industry participation, it may help to raise the confidence of the industry in stock assessment methods, monitoring capabilities, and management of our fisheries resources. The Habcam Group has also developed education activities and participated in various outreach projects. The group is currently collaborating with The Ocean Explorium, an education center and aquarium located in New Bedford, Massachusetts and local science teachers and educators.

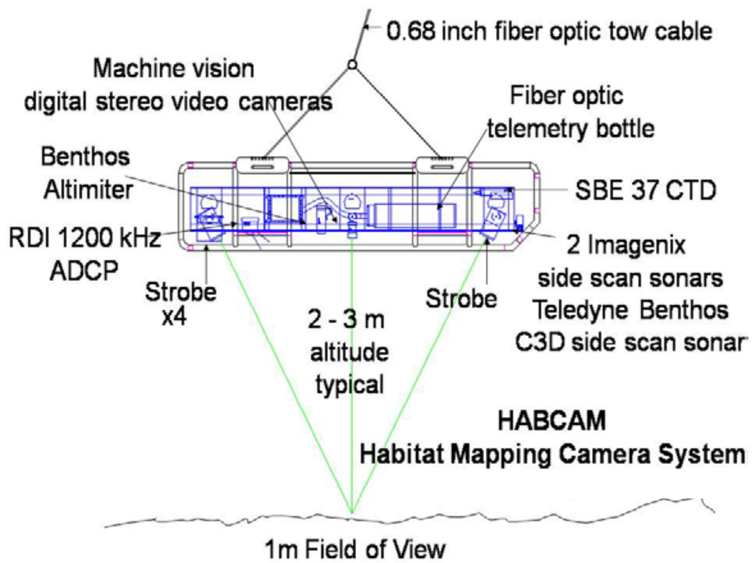
Methods and Results

Onboard sensors include a high resolution machine vision GigE color camera, four xenon strobes, side scan sonar, CTD with temperature, salinity, chlorophyll, turbidity, and pH, and a variety of engineering sensors including vehicle roll, pitch, and heading (Fig. 1). All sensors are networked subsea and data transferred via a GigE network to the surface so that data are collected and sent to the ship in real-time where they are recorded, time stamped, and stored.

The HabCam imaging system is "flown" by an operator who controls the winch keeping the vehicle 1.5 to 3 meters off bottom while being towed at 4 to 5 knots (~2.5 m/sec). A track approximately 100 nautical miles is imaged each 24 hour day while at sea. Optical imagery is collected at a width of approximately 1 to 1.25 meters (total ~200,000 m^2 /24 hr day). Images (1280x1024 pixels, 16 Bit) are acquired at 5-6 Hz providing a minimum of 50% overlap between images. Images are processed in real-time on the ship by color correcting raw 16 bit tiff images and converting them to 24 bit jpegs (Fig 2). Figure 2 represents a combination of existing data structures and what we envision as fully operational database.

The current NEFSC survey dredge is 8' wide dredge makes approximately 24, 15 min tows at 3.8 kt per day, covering about 4,500 m^2 per tow and 106,704 m^2/day. Continuous operations with HabCam towing at 5 kt and producing 5 images per second with 50% overlap covers 259,200 m^2/day. Thus, the spatial coverage of HabCam is nearly 2.5 times the area covered by the survey dredge.

We have implemented two simultaneous and complementary forms of image informatics

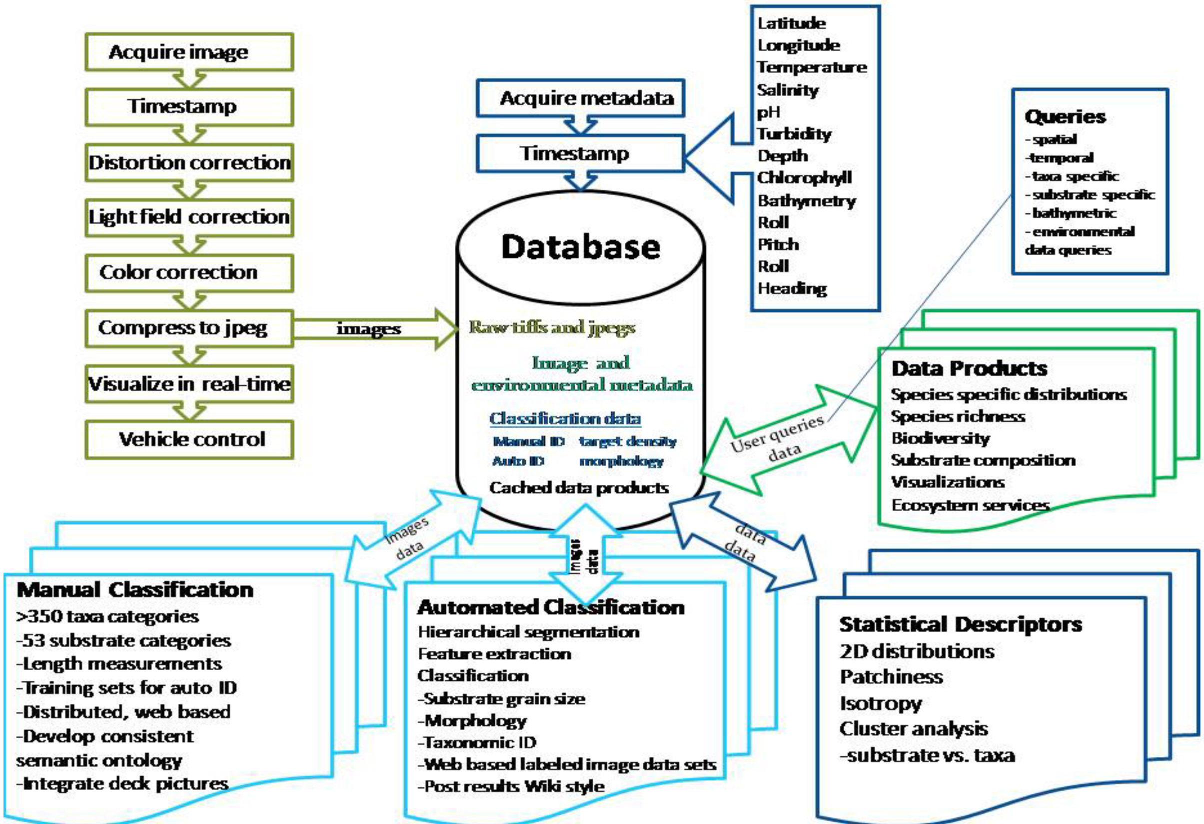


Appendix B9-Figure 1. The HabCam vehicle is towed on 0.68" fiber optic cable ~ 1.5-3 m from the bottom. The camera provides a field of view of 0.5-2 m^2. Four strobes flash synchronously with each image. Ancillary sensors include side scan acoustics, CTD, chlorophyll fluorometer, and CDOM fluorometer.

(i.e., extracting information from images): manual and automated classification. Manual

classification proceeds by having one or more operators review individual, or sets of, images to identify and measure target species using a GUI with point and click functionality. This allows about 60 to 200 images per hour per operator to be processed depending on image complexity and number of individual species being identified.

More than 460 taxa or taxonomic groups ranging in size between ~1 mm to 2 m have been observed and identified with HabCam. While taxonomic definitions used in image analysis are based on epibenthic organisms, a variety of infauna can typically be observed and quantified such as bivalve siphons, turbularian worms, burrowing shrimp, and some vertebrates (e.g., tilefish and their burrows). During manual operations, the operator also evaluates the substrate type in each image and categorizes it into one of 43 groups ranging from silt, sand, gravel, shell, cobble, boulder, and a variety of combinations. Development of approaches for automated



Appendix B9-Figure 2. Future iterations of the HabCam data workflow environment. Images and associated metadata enter the processing path from the left. Following preliminary image processing steps conducted in real-time (e.g., color correction), images are viewed and classified by scientists onboard. Results are entered into the database and used as training sets for automated classifiers. The database may be queried both spatially and temporally to build a set of data products shown on the right.

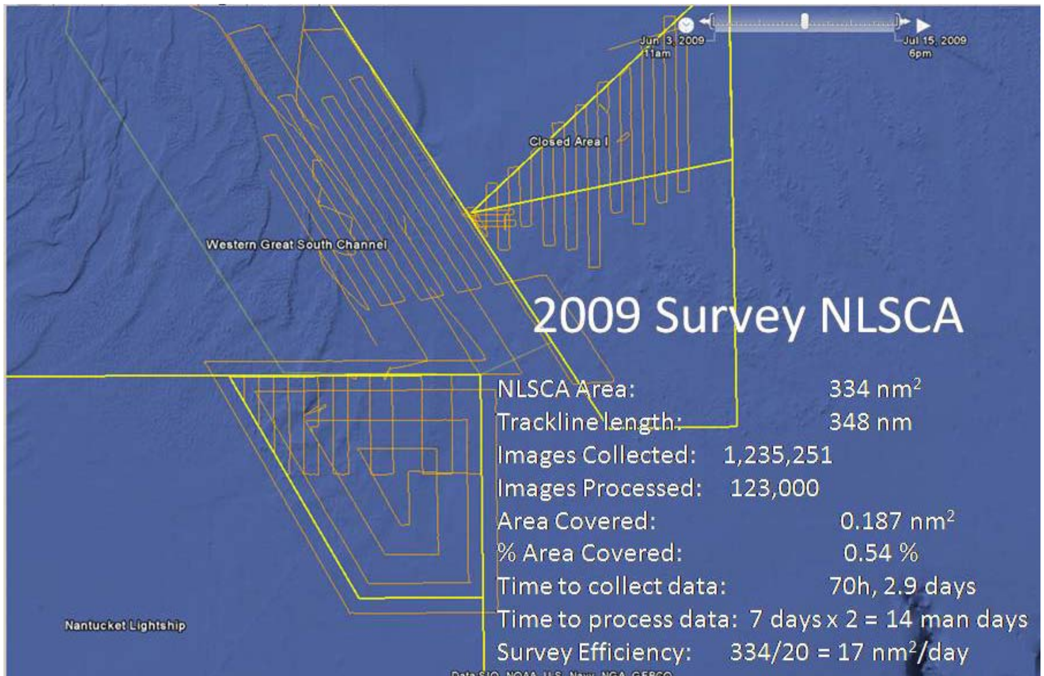
classification of targets and substrate is proceeding using the manually classified images as training sets. This is an area of ongoing research.

For the purpose of the scallop surveys reported here, images were classified manually by several technicians, who characterized substrate into the categories noted above and measured all scallops and groundfish, including yellowtail flounder. To speed the process, every 10th image was analyzed. Rationale for this strategy is discussed in the following sections.

2009 Nantucket Lightship Closed Area (NLSCA)

The objectives were to estimate scallop abundance, biomass, and shell height composition in the NLSCA using the data collected. A secondary objective was to estimate the distribution and abundance of yellowtail flounder in relation to substrate. The area of NLSCA is 334 nm^2. Total track line length was 348 nm with 1,235,251 images collected. A total of 123,000 images were processed for a total area covered of 0.187 nm^2. Total area sampled by HabCam was 0.57% of the NLSCA. The survey required 72 hours of continuous towing and approximately 27 person days to process the data, assuming 12 hour shifts.

The survey track line was a modified spiral which started 1.3 nm outside the boundary of the closed area to allow interpolation of the final abundances without boundary influences (Fig. 3). The spiral was conducted around the border then continued at an interval of 2.6 nm.

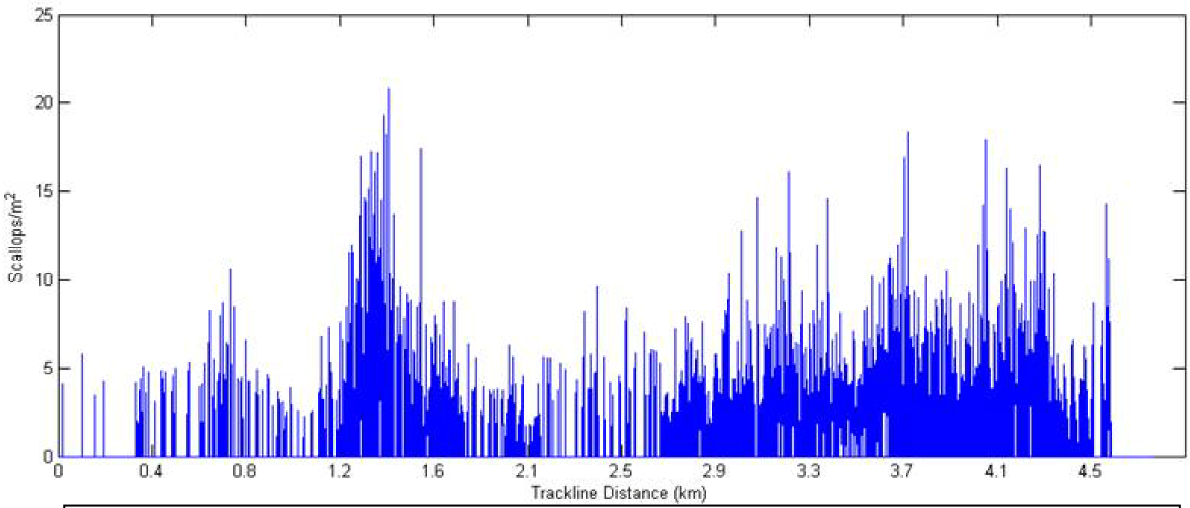


Appendix B9-Figure 3. Closed area boundaries (heavy yellow) for NLSCA and CLAI. The CLAI scallop access area is the triangle in the center of the CLAI polygon. HabCam track lines (orange) in NLSCA, CLAI and WGSC show the extent of the scallop and yellowtail founder surveys conducted in June-July 2009.

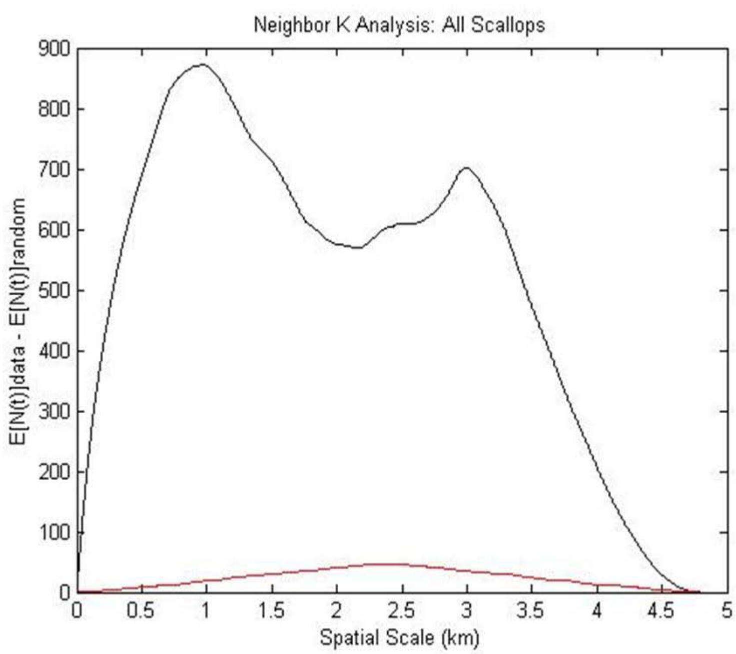
Following completion of the spiral, the vessel steamed to the northeastern corner and began a finer grid extending from the northern edge to the center of the closed area. This finer grid pattern began at an interval of 2.6 nm but compressed to 1.3 nm as the vessel approached the north-central section of the closed area. This was to provide higher resolution where prior knowledge indicated dense scallop abundance.

To assess multi-scale patchiness and to determine how many images should be processed, a preliminary transect from east to west 2 nm in length was processed by manually counting and measuring every other image. This provided information for calculating the appropriate image subsampling rate for processing the remainder of the spiral and for calculating a patchiness index for use in setting appropriate interpolation scales. Images were subsampled for the sake of efficient and fast data processing.

The density of scallops along survey tracks ranged between 0 and 23 scallops/m^2 (Fig. 4). It appeared that aggregations of scallops at densities between 5 and 20 /m^2 occurred in clumps at a spatial scale of 400m. Therefore, a Nearest Neighbor–k analysis was performed to establish the dominant spatial scales of patchiness. The Neighbor-k showed strong patchiness ranging from 400 to 900m and again at 3000m (Fig. 5).

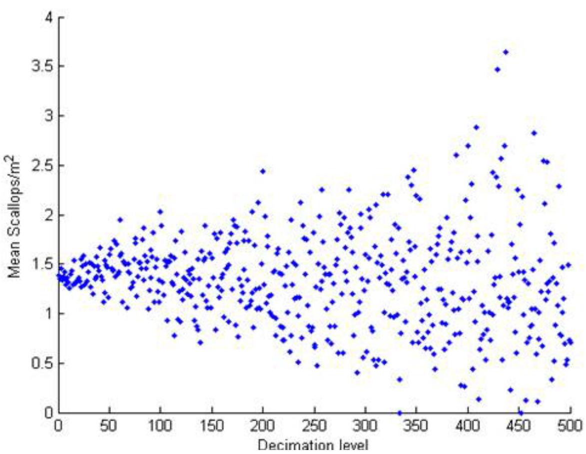


Appendix B9-Figure 4. Scallop abundance in the northern section of the NLSCA survey along a 4.5 km (2 nm) track. Every other image was classified manually to establish a baseline for subsampling and to estimate patchiness. Note the very patchy distribution ranging from 0 to >20 scallops per m^2.

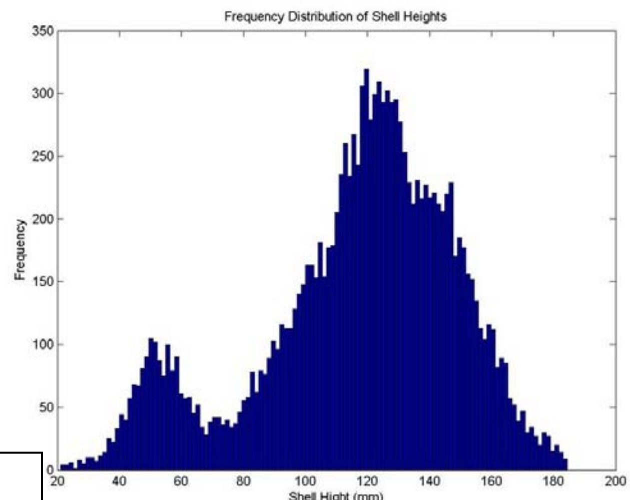


Appendix B9-Figure 5. Ripley's Neighbor-k analysis for one-dimensional patchiness was applied to scallop densities measured along the track (every other image was processed for this analysis). Spatial scale is on the x axis while the residual between observed and predicted nearest neighbor distance under randomness (1000 Monte Carlo simulations) is on the y axis. The first mode indicating a characteristic patch size is located at 700-900 m, and a second at 3.2 km. Both modes are well above the red line under randomness indicating these patch dimensions to be statistically significant at the 95% confidence level.

To determine an optimal image subsampling rate, data from every other image along the track was processed by extracting abundances and calculating the CV for density at sampling intervals ranging from every 4th, 8th, 10th, 12th etc. out to every 500th image. The mean and CV remained stable up to a subsample level of every 10th image. Therefore, the remaining spiral was processed at a rate of every 10th image (Fig. 6).
Over 12,900 scallop shell heights were counted and measured using MIP (Manual Identification Program developed by A.D. York), which allows users to quickly point and click on scallops to extract measurements and select substrate type from a menu. The shell height distribution was strongly bimodal with modes at 53 and 125 mm (Fig. 7). A third, less prominent mode was located at 142 mm.



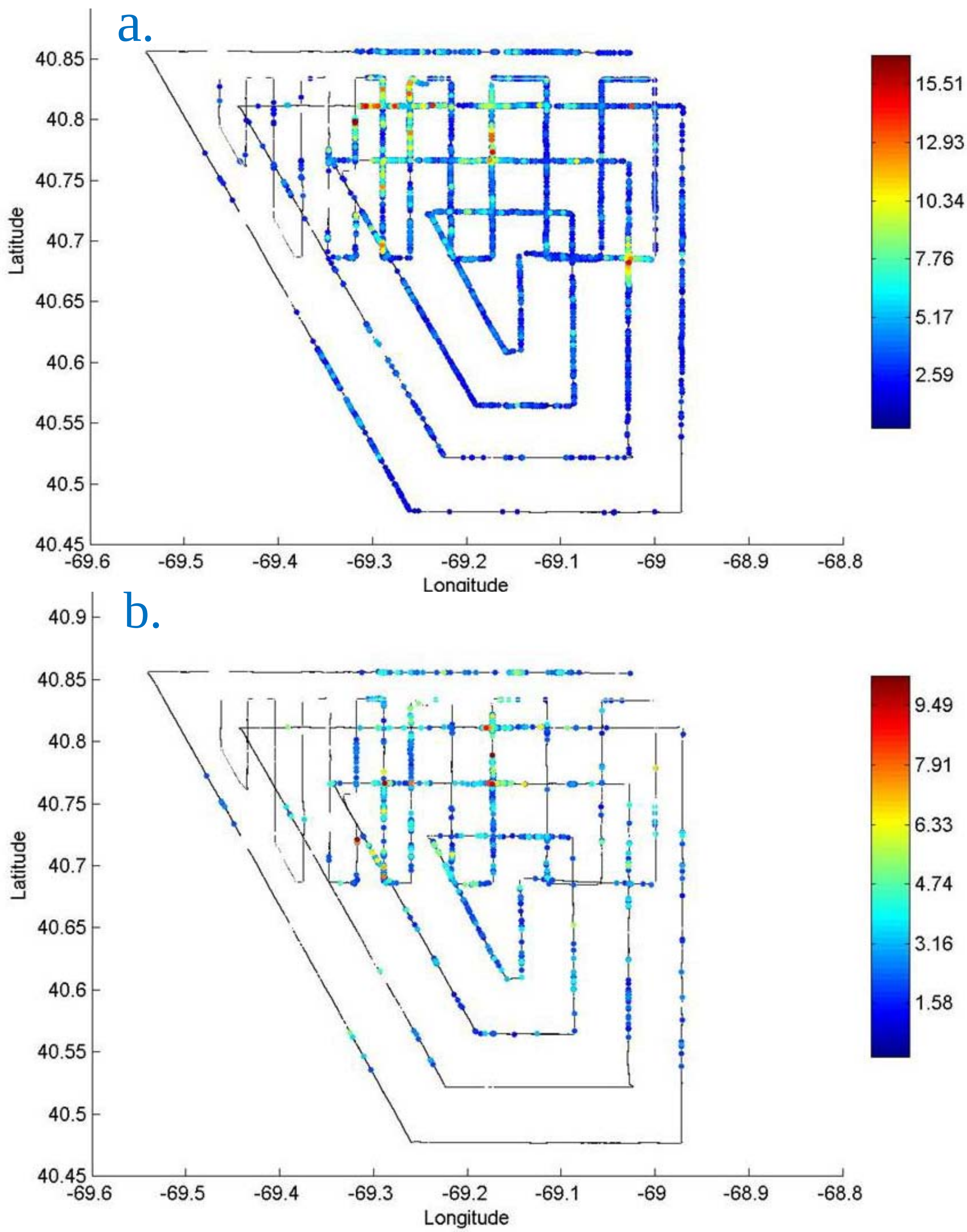
Appendix B9-Figure 6. The effect of subsampling (decimation) the continuous record of scallop abundance along the track line in Figure 3. Images were analyzed at subsampling rates of every 4th, 8th, 10th, 12th… out to 500 and the scallop abundance recalculated. Mean abundance is stable out to a subsampling rate of greater than 20 so a conservative level of processing every 10 image was chosen for the remainder of the analysis.



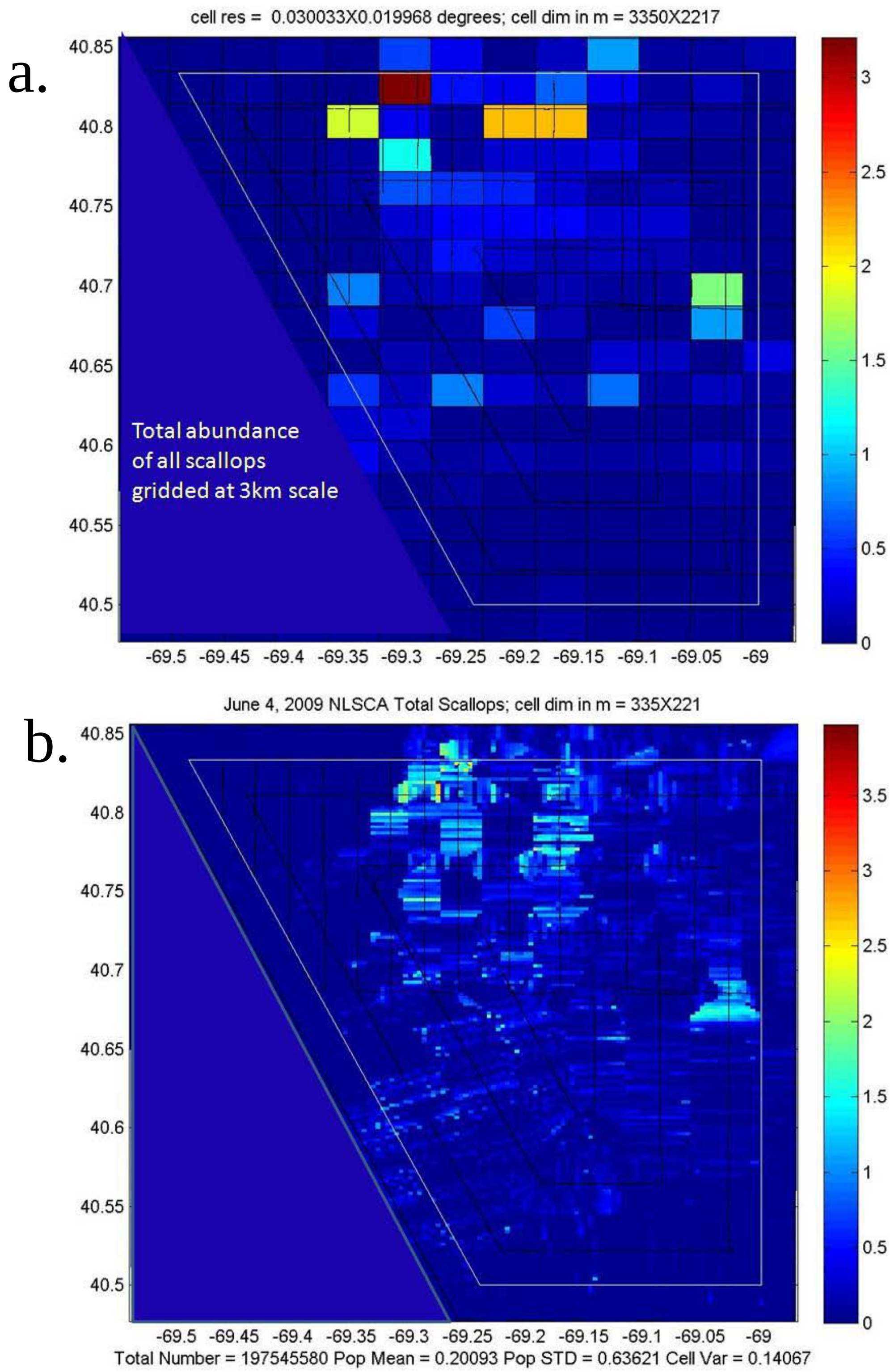
Appendix B9-Figure 7. Shell height frequency distribution for all scallops measured from the NLSCA survey. N=129,237. The shell height distribution was strongly bimodal with modes at 53 and 125 mm with a third, less prominent mode located at 142 mm.

The scallop density from every 10th image plotted as color coded dots for all scallops showed highest aggregations in the central upper third and in the central eastern region of the closed area (Fig. 8a). Scallops were sparse in the northwestern corner and southern regions. A similar plot for just those scallops with shell height between 20 and 65mm showed that small scallops were most abundant in the east central region of the closed area (Fig. 8b).

The raw scallop density per image was interpolated into rectangular grids at two scales using ordinary kriging based on cells of 3350x2217m and 335x221m. First, a semi-variogram was constructed to evaluate autocorrelation of the data. For the coarse and fine scales, the mean for each grid cell was color coded (Fig. 9a, b). Total abundance, mean, and variance for each grid cell were calculated. An overall CV was calculated by bootstrapping the standard error divided by the mean for each grid cell. Data were collected and kriged beyond the location of the closed area boundary, but results presented are only for the area within the NLSCA boundaries. The highest mean value was 4 and 6 scallops/m^2 for the coarse and fine scale grids, respectively.

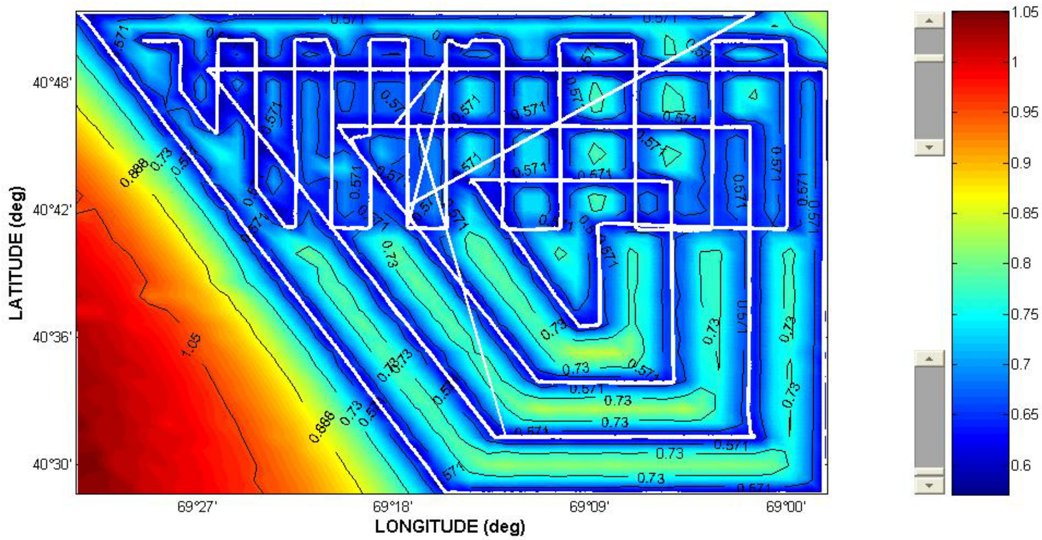


Appendix B9-Figure 8. Raw abundance (#/m^2) estimates on a per image basis for all scallops regardless of shell height. Each dot represents a single image with the abundance indicated by color. Where no dots exist, no scallops were observed. a) All Scallops (#/m^2). b) Scallops with shell height less than 60mm.



Appendix B9-Figure 9. a) Kriged scallop densities at a scale of 3350x2217 m . Mean densities represented by color referenced to the color bar on the right. b) Kriged scallop densities at a scale of 335x221 m showing considerable patchiness at this fine

Total abundance within the closed area boundary was calculated by summing each of the grid cells that fell within the boundary. Those cells that were partially within the boundary were evaluated by including only the proportion of the cell falling within the boundary. Total scallops in the coarse and fine scale grids were 174,966,666 and 197,545,580 scallops, respectively. The discrepancy in total abundance between grid scales probably lies in the fact that the scallops were patchy at scales of 400-900 m as shown from the Nearest Neighbor-k analysis. The courser grid scale smoothes high density patch values over a larger area than the higher resolution grid. The finer grid, therefore, is providing a more representative view of the scallop distribution and also the most accurate estimate. The overall mean for the fine scale grid was 0.187 scallops/m^2 with a CV of 0.04. Variance between cells ranged between 0.5 where sampling density was greatest to over 0.7 where sample density was low (Fig. 10).



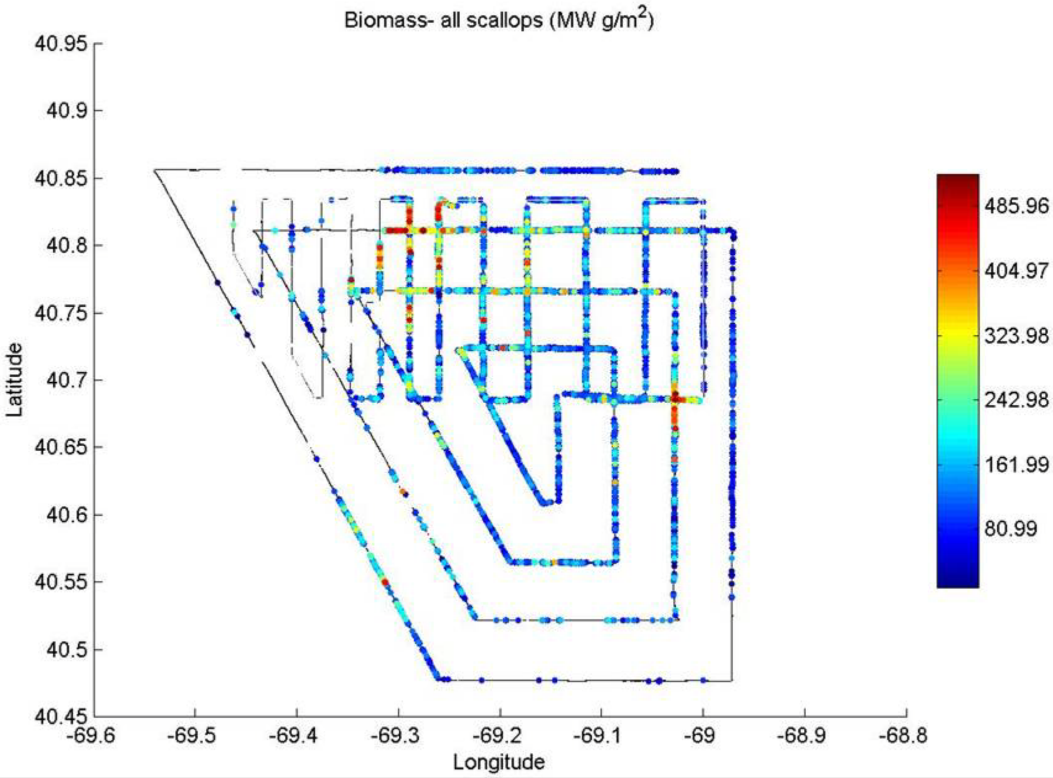
Appendix B9-Figure 10. Variance per cell for kriged scallop densities at the fine scale.

The weight of individual scallops used to estimate biomass was calculated using a shell height-meat weight relationship that included depth:

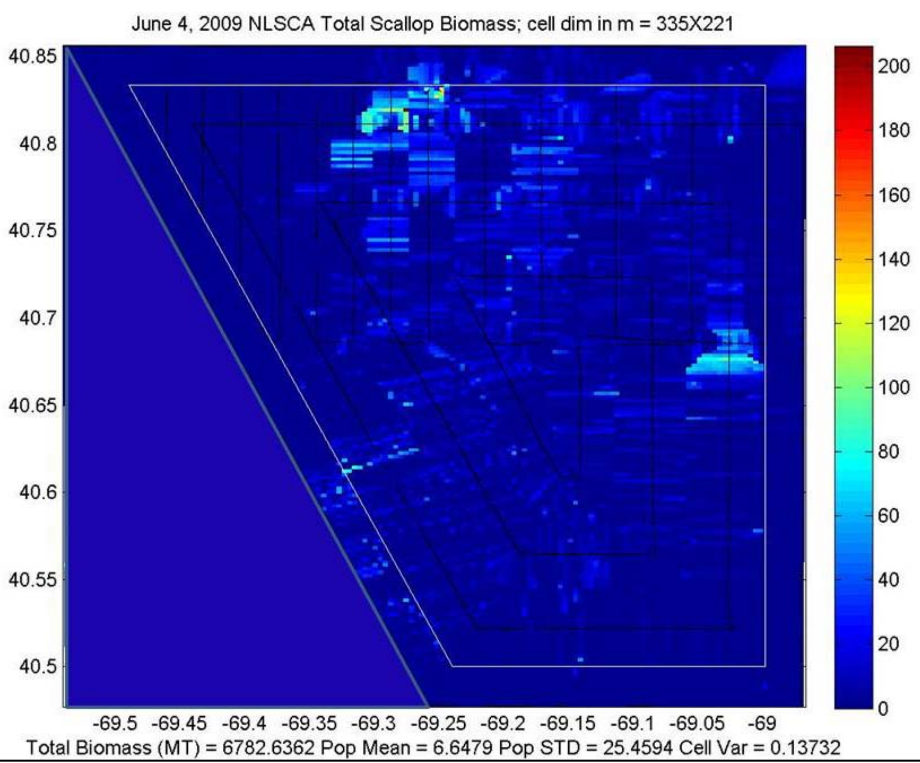
$$W = \exp(a+b \log(SH) + c \log(depth)$$

where W is weight (g), SH is shell height (mm), depth is in meters and the parameters a = -8.62, b = 2.95, and c = -0.51 (D. Hart, NEFSC, pers. comm.).

Mean biomass per scallop was 32.9 g. The basic pattern of distribution followed that of scallop abundance with dense scallop areas in the central northern region and in the central eastern region (Fig. 11). Biomass estimated by kriging at the fine scale 6,782 MT meats (Fig. 12).

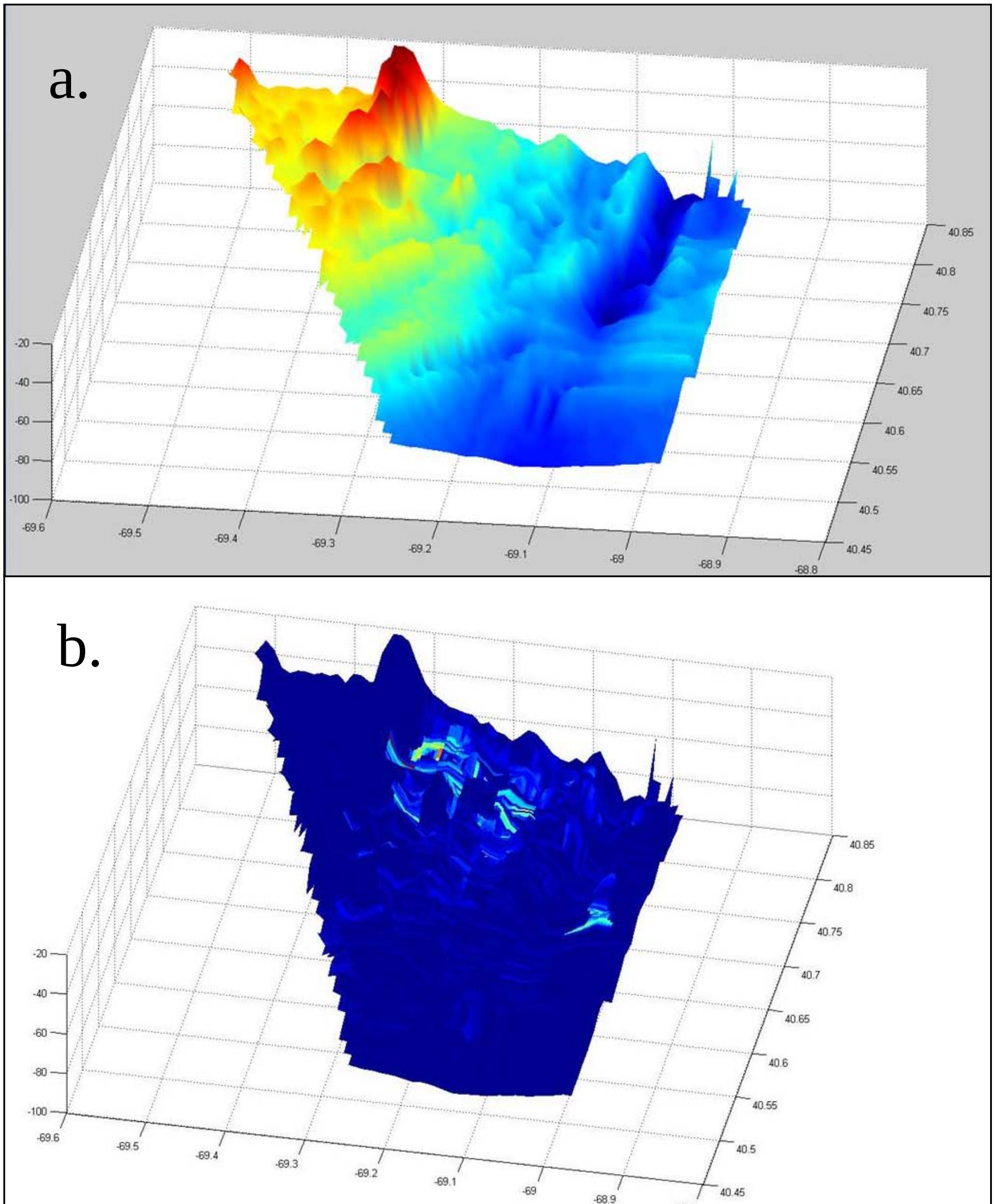


Appendix B9-Figure 11. Scallop biomass densities along the track line for all scallops of all sizes.

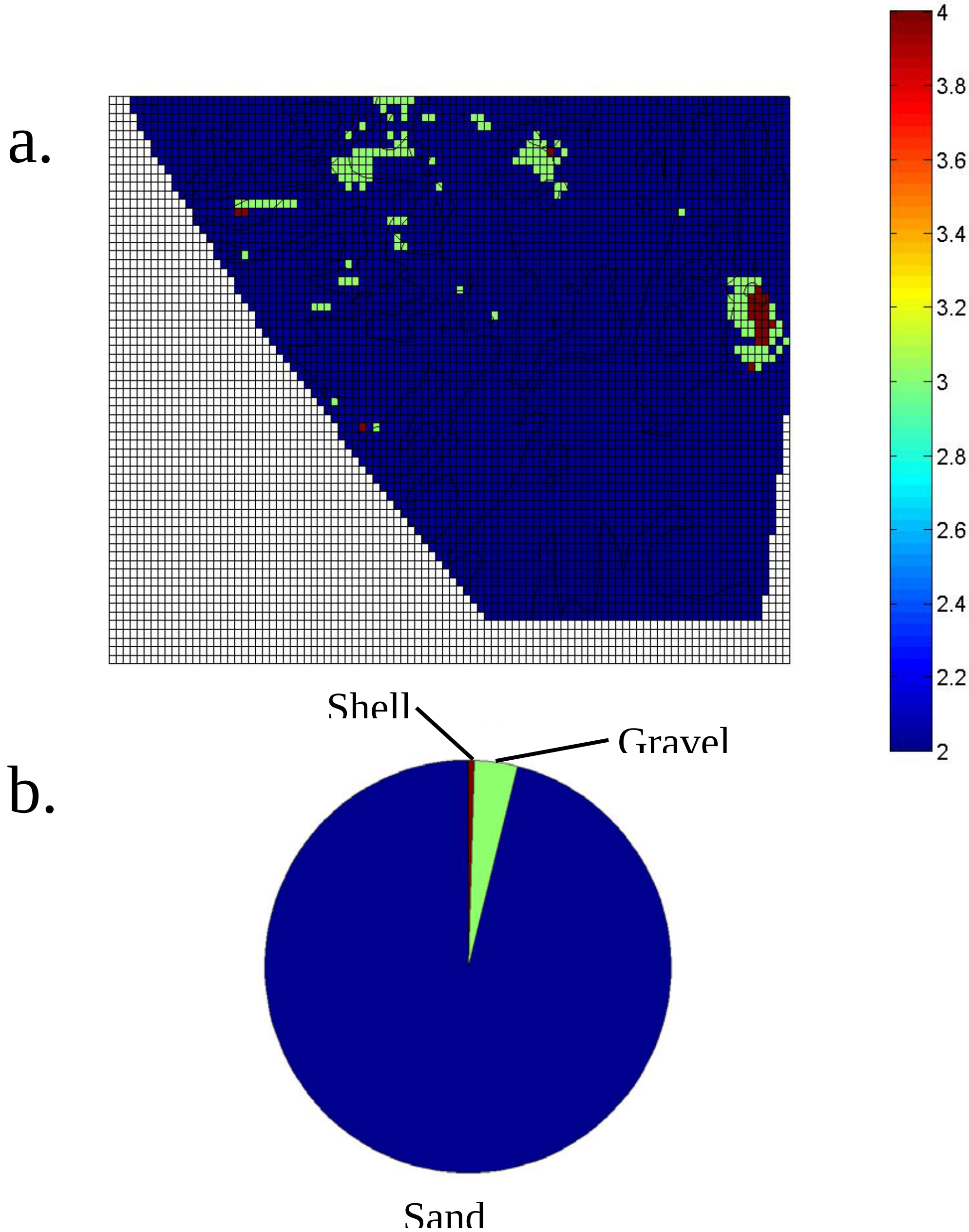


Appendix B9-Figure 12. Kriged biomass estimates generated at the fine scale for scallops of all sizes.

To examine relationships between density of scallops, depth, and substrate type, the depth from the ships echosounder was linearly interpolated onto a uniform grid and colored as a function of depth (Fig. 13a). Sand waves in the northern central region and a trough in the central eastern region are notable. When scallop density was plotted as a color map over the interpolated depth data, it was clear that greatest densities were at the eastern base of the sand waves and just eastward of the trough, but not in the trough (Fig. 13b).

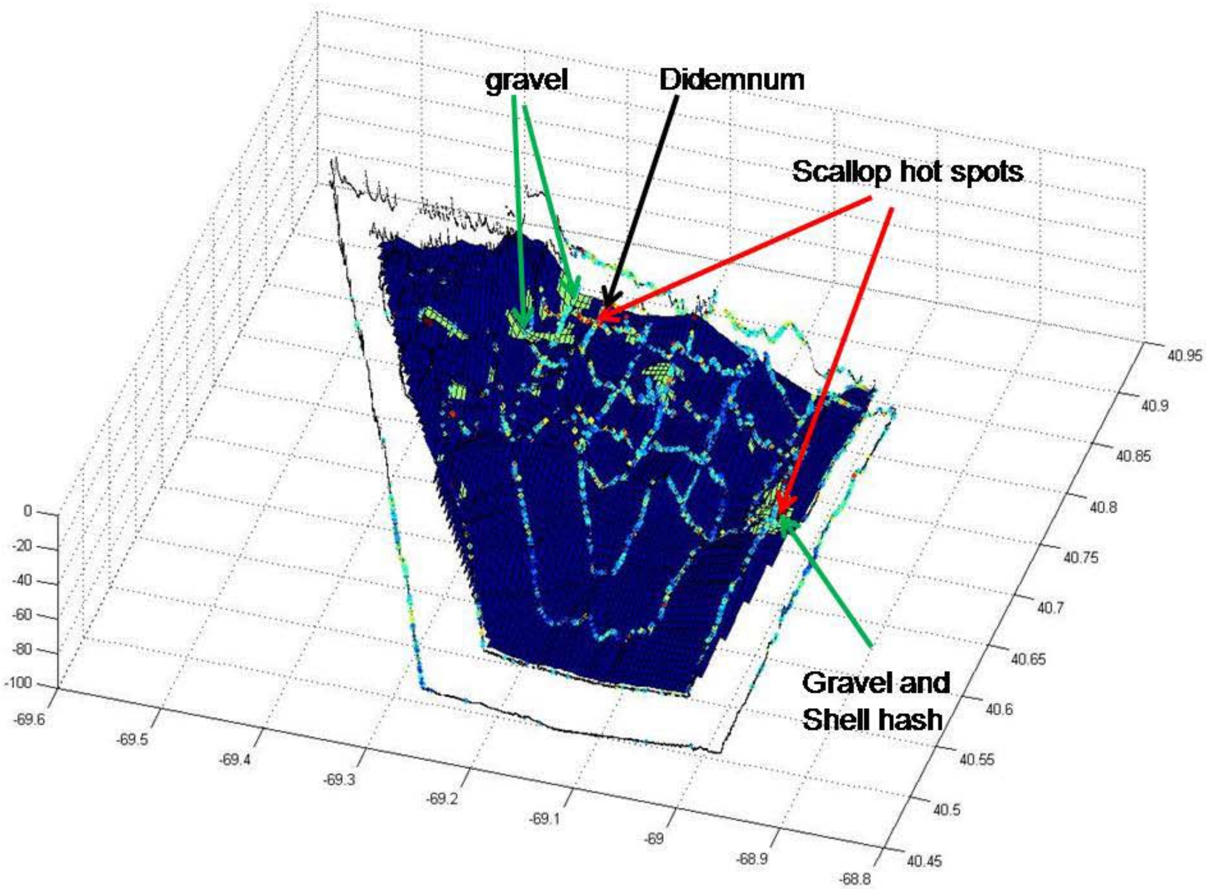


Appendix B9-Figure 13. a)Depth from the ship's sonar interpolated to a uniform grid.
b) Depth with overlaid scallop abundance on the same color scale as in Fig. 12.
Z axis is exaggerated for visualization purposes.

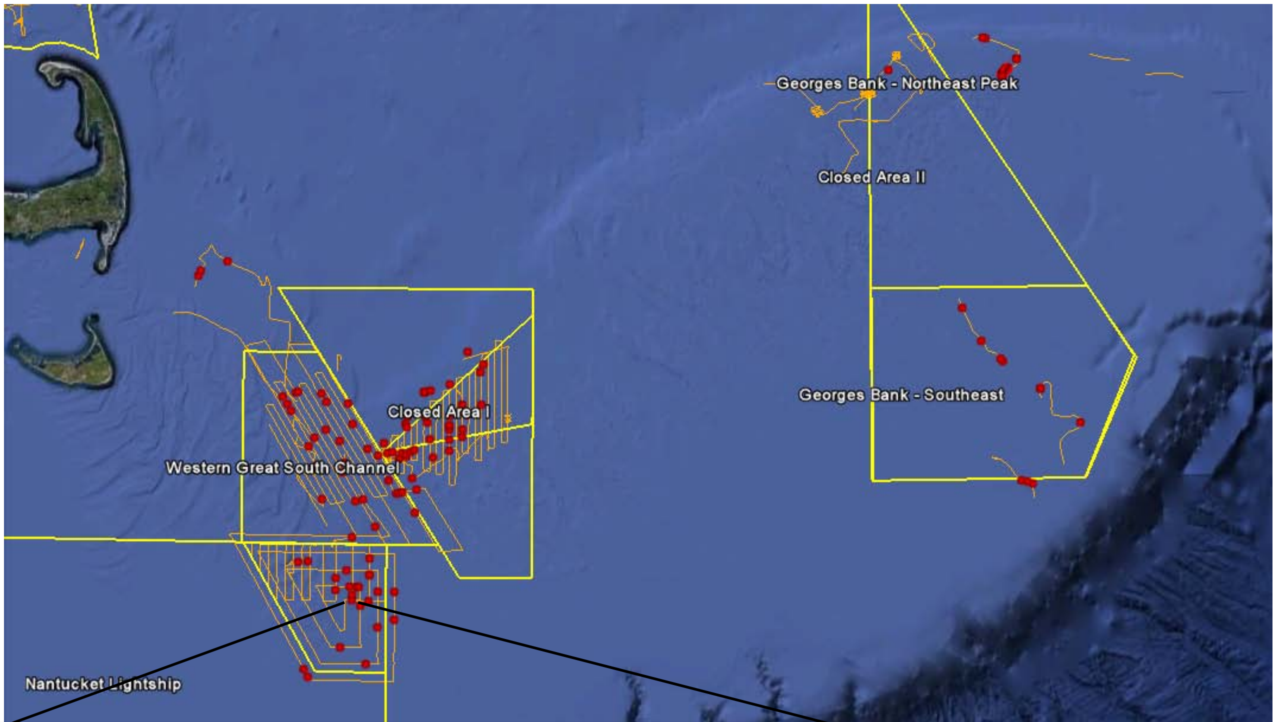


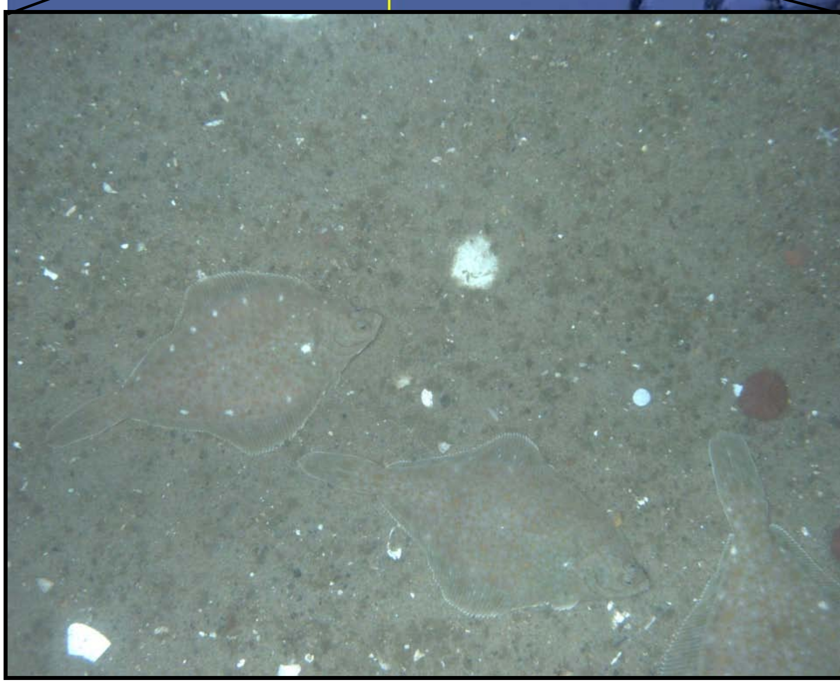
Appendix B9-Figure 14. a) Dominant substrate binned numerically into three categories (2) sand, (3) gravel, (4) shell. b) Sand dominated pie chart of substrate in NLSCA.

Substrate classifications were re-categorized into three numeric bins of dominant substrate: (2) sand, (3) gravel, and (4) shell. Dominant substrate categories include mixed substrate types, for example, "gravel" contains mixed substrate images such as gravel/sand and gravel/shell. Interpolation of these substrate categories across the NLSCA grid showed the entire area to be mostly sand (Fig. 14a,b). The greatest accumulation of sand/shell hash corresponded to areas of high scallop densities. The region to the central eastern side of the trough had notable sections of gravel, which is also where scallops were most abundant, particularly scallops less than 60mm in height. The combination of all three variables, scallop density, depth, and substrate (Fig. 15) provides a visualization of how scallop distribution is affected by these variables. HabCam data for the invasive tunicate *Didemnum vexillum* collected simultaneously with sea scallop data during the survey, illustrate spatial relationships of two species and substrate type and demonstrate the potential for use of data in ecological studies (Fig. 15).



Appendix B9-Figure 15. Depth overlaid with scallop abundance and substrate. Note location of invasive tunicate *Didemnum vexillum* in relation to high scallop densities.



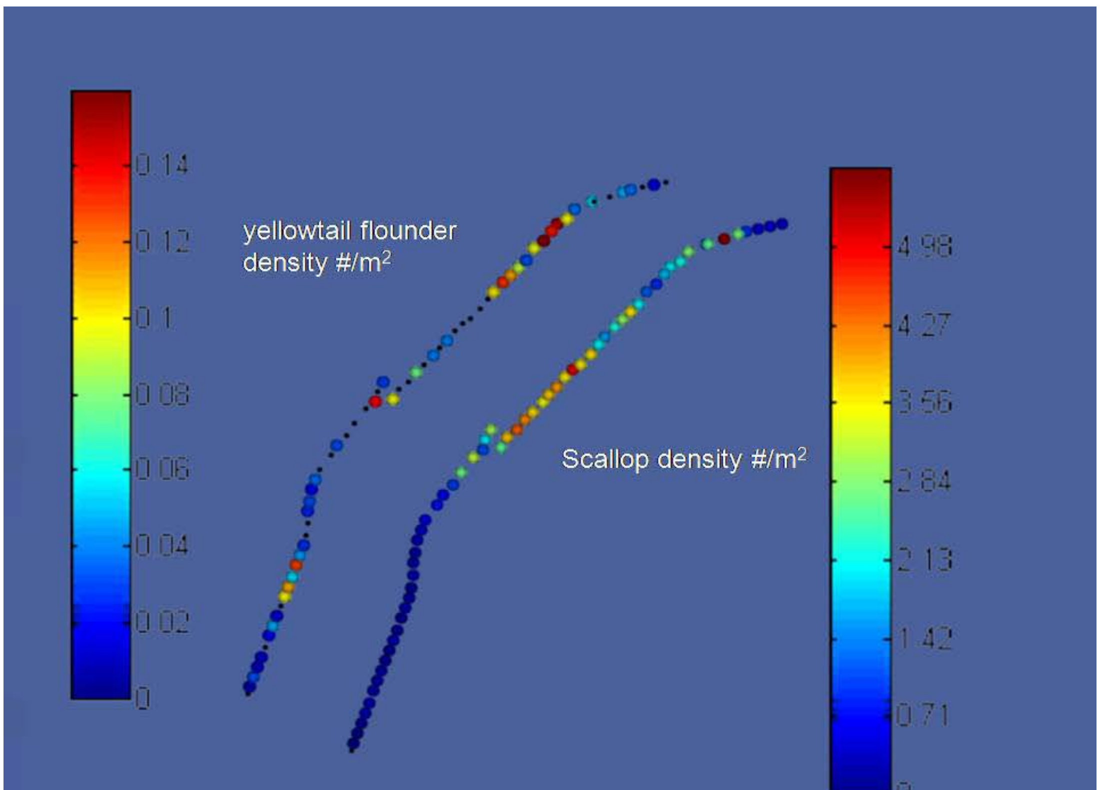


Appendix B9-Figure 16. (top) Observations of yellowtail flounder in CLAI, WGSC, NLSCA, CLAII and on the Canadian side of the Northeast Peak. Each red dot represents an individual sighting. (bottom) Example of three yellowtail in a single image, taken in NLSCA.

As an alternative method for calculating total population abundance of scallops without kriging or interpolating, one may simply use the overall mean observed in images multiplied by the total area. Our results using this approach is 0.187 scallops/m^2 x 1,142,280,000 m^2 = 213,606,360 scallops with a CV of 0.034.

In addition to sessile organisms, the HabCam system may be useful for imaging mobile demersal fishes. Yellowtail flounder were observed in NLSCA and other regions during our survey at relatively low densities (Fig. 16). In NLSCA, 124 observations were made with the densest concentration in the central region. This region was also characterized by being mostly sand with patches of gravel. The most abundant aggregations of yellowtail were observed in the Southeast Part of CLAII and on the Canadian side with densities exceeding 0.14 fish/m^2. In some cases two or three fish were observed in a single image. Images from CLAII show yellowtail to be found on mostly sandy bottom with shell hash and occasionally on gravel.

An interesting relationship between scallop and yellowtail density was observed on the Canadian side of the Northern Edge. Data for yellowtail and scallops for the same track line are plotted alongside each other in Figure 17. Note that yellowtail appeared to be at highest densities where the abundance of scallops were low. This seemingly inverse relationship only holds for this track line at that point in time and is probably related more to substrate, food supply, reproduction, or environmental variables, than a true relationship between scallops and yellowtail. These results indicate the potential of HabCam data for use in fisheries management where, for example, the goal is to reduce bycatch of yellowtail during scallop fishing.



Appendix B9-Figure 17. Yellowtail flounder and scallop densities observed along a 1.7 nm long track line on the Canadian side of the Northern Edge. Each place mark represents data binned at 50 m intervals.

Joint ship operations

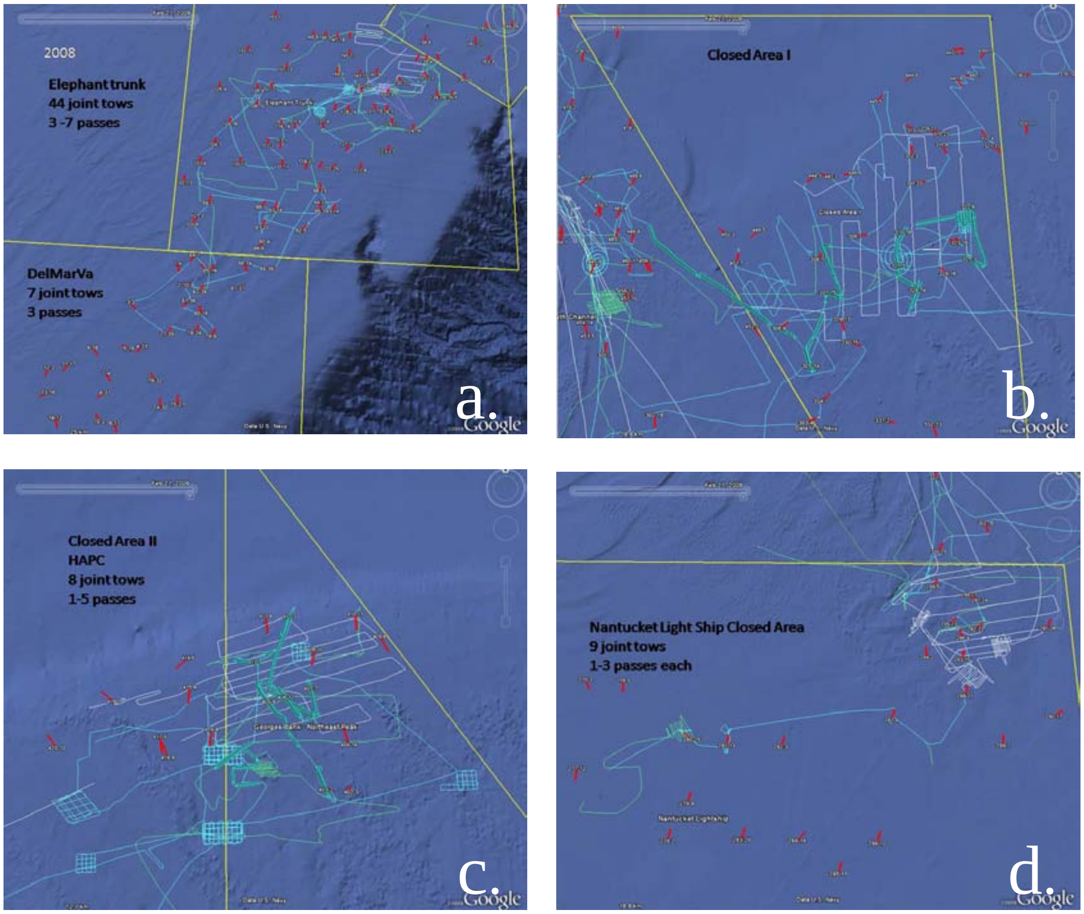
Since 2007, The HabCam Group has been collaborating with the NMFS in their annual scallop surveys by conducting paired tow experiments. These joint tows were designed to compare scallop abundances and size estimates from the standard federal dredge survey with those derived from HabCam imagery. Data will be presented here for 2008 and 2009.

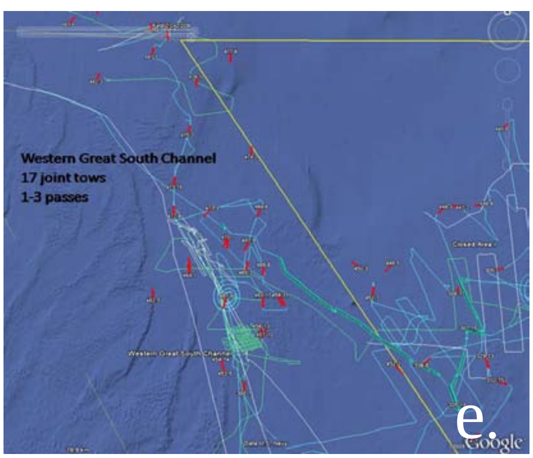
In June and July 2008, The F/V Kathy Marie 'shadowed' the R/V Sharp on 113 total tows with 44 in the Elephant Trunk, 35 in CLAI, 8 in CLAII HAPC, 9 in NLSCA, and 17 in the proposed WGSC HAPC (Fig. 18). HabCam made at least three passes at over 50% of the NMFS stations, and in a few cases made up to seven. These multiple passes were designed to assess the variability of scallop density along each track and between multiple passes. Images from all passes were processed at a subsampled rate of every 10th image. This translates into processing about 1 m^2 for every 5 m of track line.

Within hours of conducting a dredge tow, the beginning and end points for the tow were communicated at sea via radio from the Sharp to the Kathy Marie. This allowed the captain of the Kathy Marie to line the vessel up on the dredge tow and follow a straight line from one end to the other. Because the absolute position of neither the dredge nor the HabCam vehicle was precisely known, our best efforts were to make multiple passes that coincided within about 50m of the dredge tow line and between each pass of HabCam. As an example, data for seven passes along the dredge tow for one station in Elephant Trunk (91) shows within and between variability of scallop densities observed by HabCam (Fig. 19). Although Pass 6 appears to be an outlier, results of a one-way ANOVA suggest that there is no significant difference between all 7 passes ($p<0.001$).

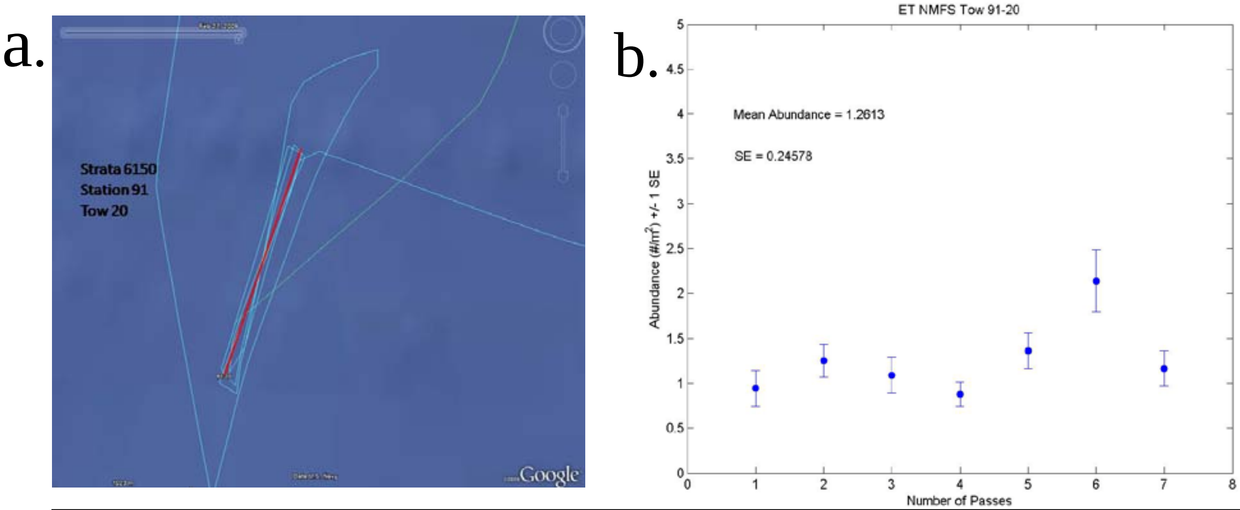
HabCam estimates of scallop abundance were consistently greater than dredge counts, indicating that dredge efficiency is well less than 1. Mean shell height measurements were similar between dredge and HabCam (Fig. 20), but as will be discussed in the section under error analysis, the tails of the frequency distribution for HabCam are higher than those for the dredge indicating some inherent error in the measurement of shell heights. Count data tend to be accurate in optical surveys but some degree of body size measurement error is typical (Jacobson et al., 2010). This is an area of ongoing research.

In June 2009, in addition to shadowing the Sharp with the Kathy Marie during Legs 1 and 2 of the annual scallop survey, HabCam was towed from the A-frame of the R/V Hugh R Sharp as part of routine dredge operations on Leg 3. This project was designed for comparison of HabCam data for sea scallops and yellowtail flounder with data from the standard dredge tows during Leg 3 of the 2009 NMFS Scallop Survey. Because of sea state and time considerations, HabCam was towed at and between 23 stations. HabCam collected a total of 787,832 images with a footprint of about 1 m^2 each. By area, 85,572 images were collected in CLAI, 216,809 images in CLAII, 183,070 images on the Canadian side of the Northern Edge of Georges Bank, and 302,381 images between stations. A final report has been filed with the NOAA CINAR office and Russell Brown at the NEFSC (HabCam Group, 2010).

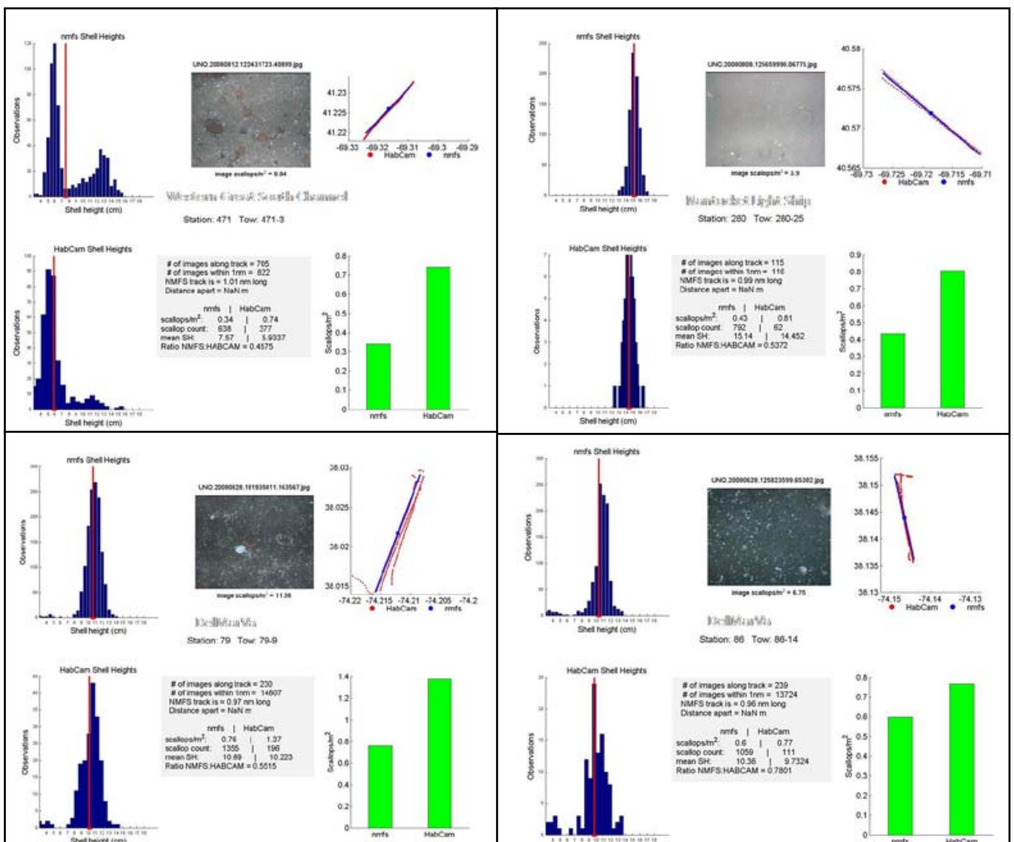




Appendix B9-Figure 18. 44 Joint tows between R/V Sharp and HabCam on the F/V Kathy Marie in the (a) Elephant Trunk, (b) CLAI, (c) CLAII HAPC, (d) NLSCA, and (e) WGSC. Red lines are dredge tows, blue lines are HabCam track lines.



Appendix B9-Figure 19. (a) Federal dredge station 91 in ET with 1 nm tow shown in red. Seven passes by HabCam shown in blue. Multiple passes of HabCam were within 50m of each other. (b) Mean +/- SE of scallop abundance (#/m^2) from each of the seven passes at station 91.



Appendix B9-Figure 20. Data from four joint stations illustrating the relationship between dredge and HabCam data. In each of the boxes shell height frequency distributions, mean abundance, and position along track for the dredge and HabCam are compared.

As each image represents about 1 m^2 and there is approximately 50% overlap, an area of about 242,000 m^2 was imaged. In an area on the Canadian side called the ‘seed box’, the density of small (50-60 mm) scallops was extremely high, upwards of 50 to 90 scallops per image (e.g. Fig. 21).

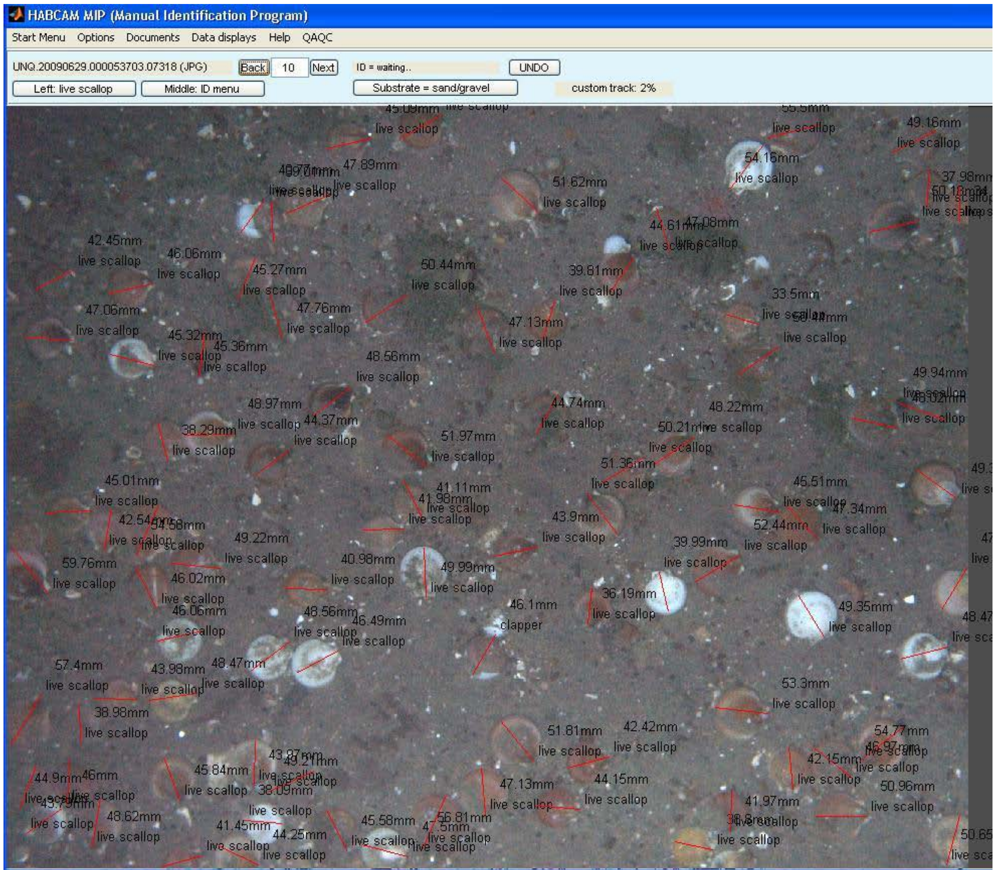
Shell height measurements from HabCam showed a strongly skewed distribution to the left with a mode of 55 mm and a mean of 79 mm (Fig. 22), indicating that this area was dominated by two year old scallops with relatively few older individuals.

Along track abundance of scallops at Station 404 ranged from 0 to well over 60/m^2 (Fig. 23). A Neighbor –k analysis of scallop distributions along the track in Fig. 4 showed that patchiness was significant at several spatial scales from 600 to 1000m (Fig. 24).

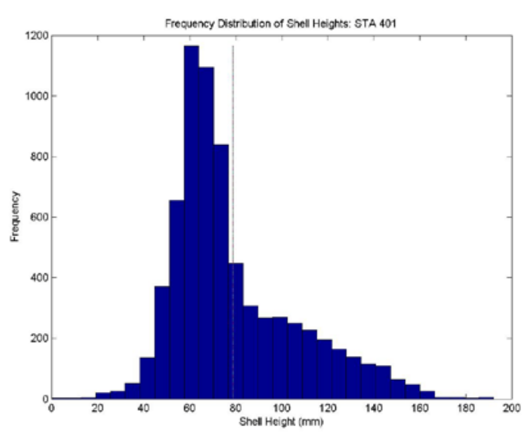
Yellowtail flounder were sparse but most abundant in the Southeast Part of CLAII and on the Canadian side of the Northern Edge (Fig. 25). Images show yellowtail to be found on mostly sandy bottom with shell hash and occasionally on gravel (Fig. 26).

Survey dredge capture efficiency is low relative to optical surveys and might be variable due to tow direction in relation to tidal currents, substrate composition, wire out, tow speed, and tow duration. To compare scallop abundances estimated by the NMFS dredge and HabCam, plots were generated by region and by substrate and include data for both 2008 and 2009 (Fig. 27). Georges Bank includes NLSCA, WGSC, CLAI and CLAII. Mid Atlantic Bight includes Elephant Trunk, Delmarva, and Hudson Canyon. Regression slopes were 0.34 for Georges and 0.46 for Mid Atlantic Bight. When the data are broken out by substrate regardless of region, the regression slope for sand was 0.35, for sand plus other substrate types such as shell hash it was 0.40, and on gravel it was 0.35. These slopes should modestly underestimate the sampling efficiency of the dredge relative to HabCam (due to errors in variables, i.e., that the x coordinates in the regression are uncertain since HabCam does not go over the exact same ground as the dredge). Results (dredge sampling efficiencies for sea scallops ~ 0.3 to 0.45) are similar to results from other studies. Moreover, they illustrate the potential for use of HabCam in directly estimating the sampling efficiency of other types of survey and fishing gear. Estimates from simple regressions are biased low because of errors in variables: see Appendix X for unbiased methodology.

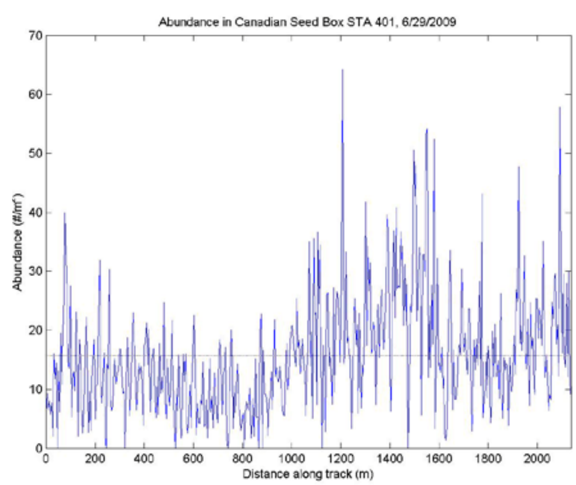
Bland-Altman plots are used to assess the correspondence between two forms of measurement for the same data and are constructed by plotting the differences between paired observations from two data sets against their mean. It was necessary to normalize the residuals for the sum of both dredge and HabCam samples. The mean residual for all data for 2008 and 2009 was 0.37 (Fig. 28), which is consistent with the regression analyses presented in Figure 27. The residuals are normally distributed between the limits of agreement suggesting that while there is a strong systematic bias, neither measurement approach is affected by abundance of scallops being measured.



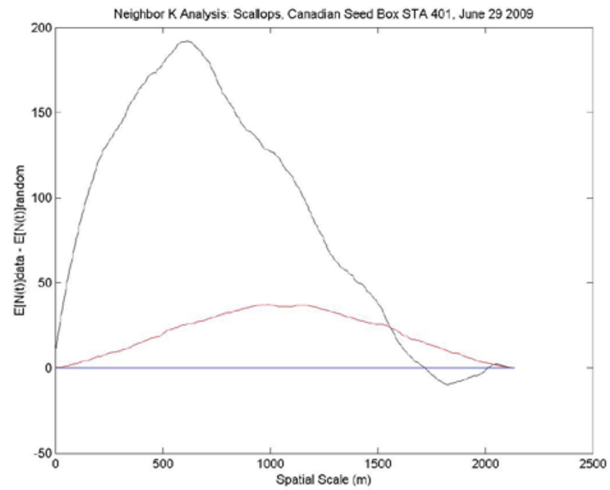
Appendix B9-Figure 21. The HabCam Manual Identification Program (MIP) while processing an image collected on the Canadian side near station 404 where 90 scallops were counted and measured in a single



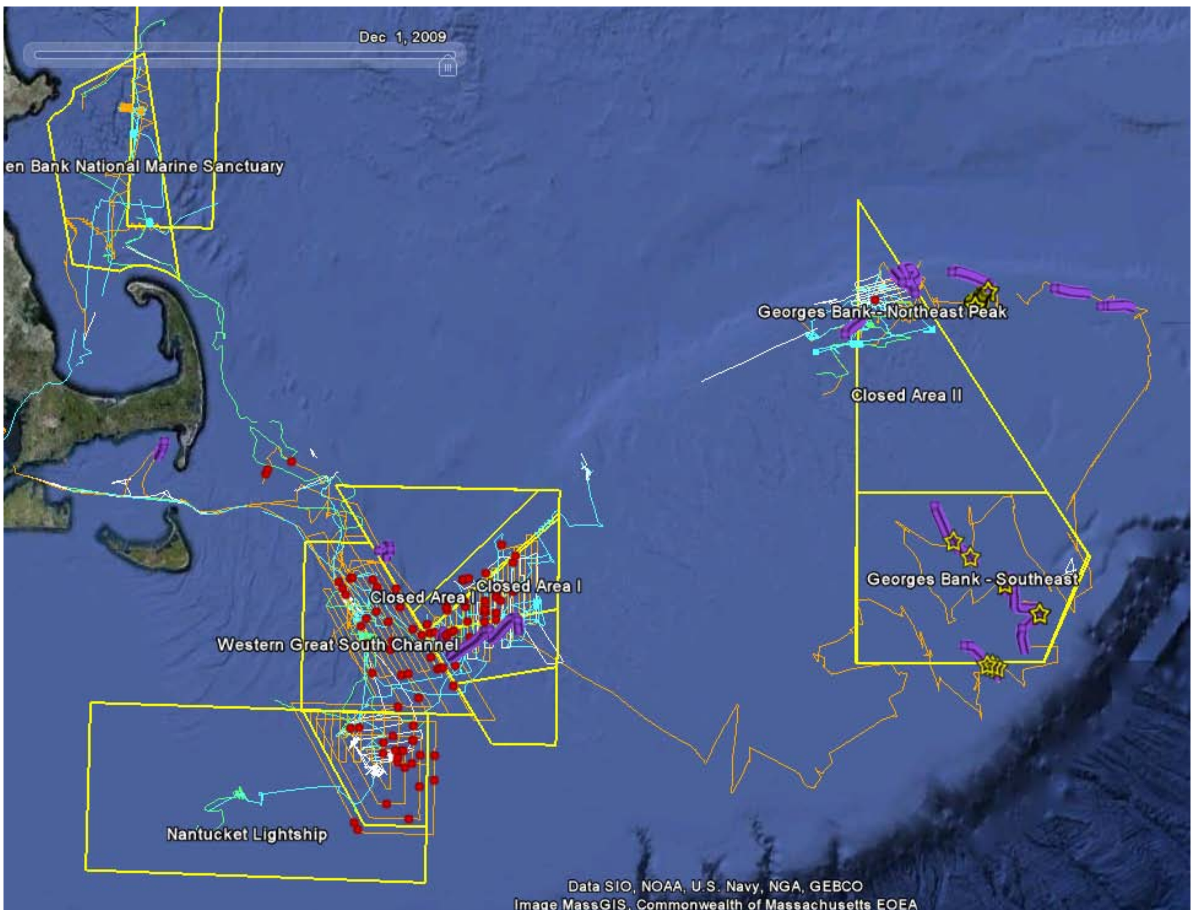
Appendix B9-Figure 22. Frequency distribution of scallop shell heights from HabCam images at Station 401 in the seed box.



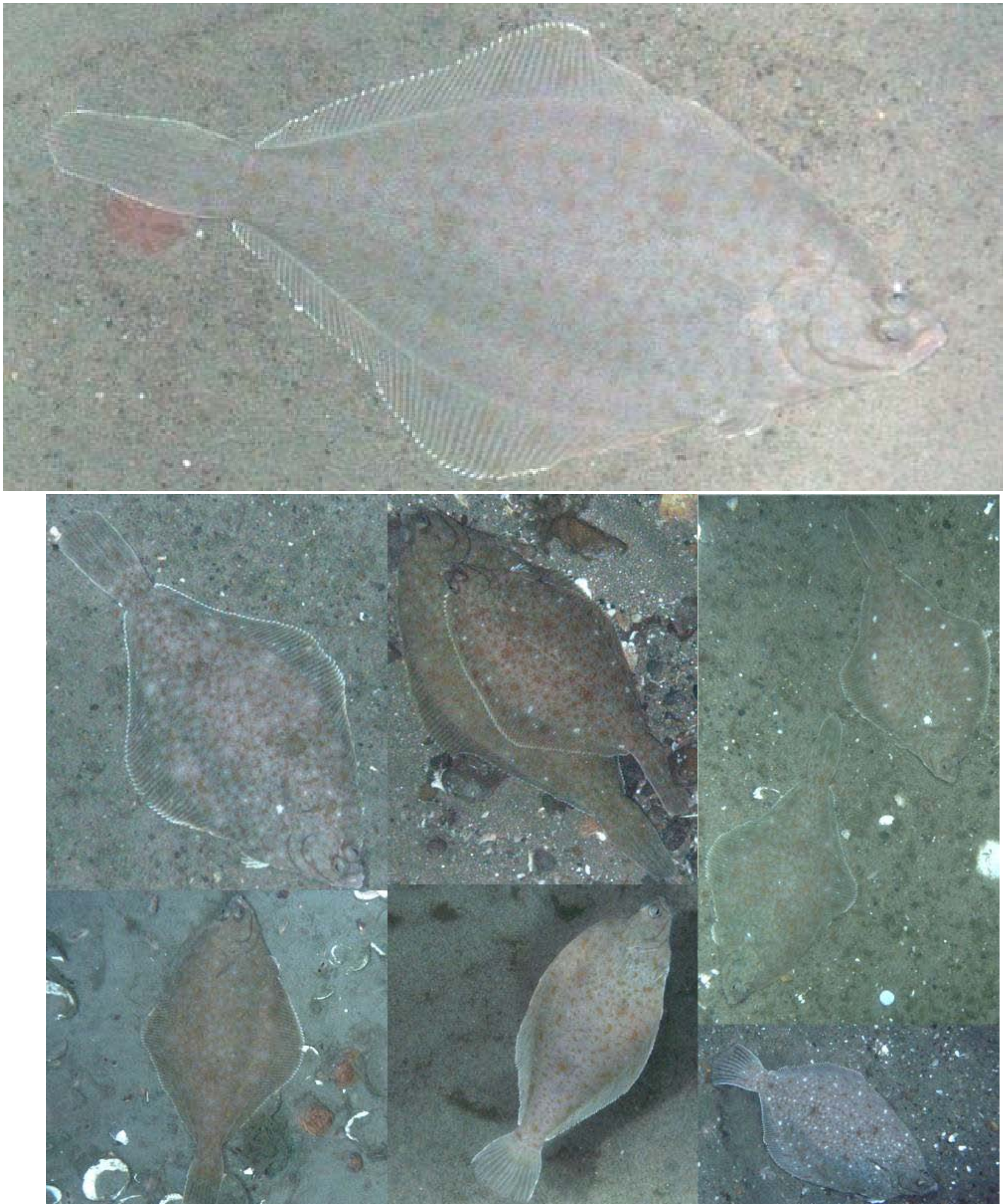
Appendix B9-Figure 23. Along track abundance of scallops at Station 401 in the 'seed box' on the Canadian side. Mean abundance was 16 scallops/m^2.



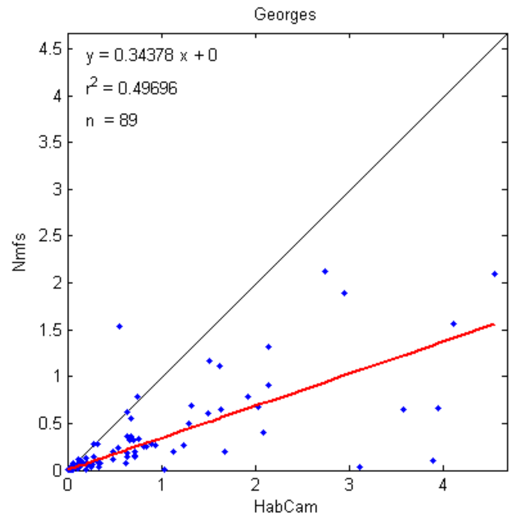
Appendix B9-Figure 24. Neighbor-k analysis of the distribution of scallops at Station 404. Note significant

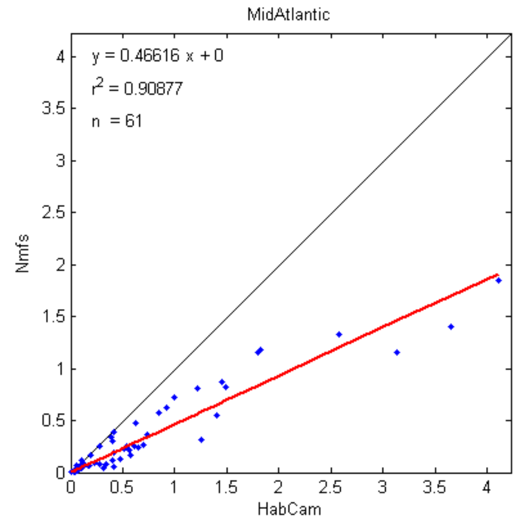


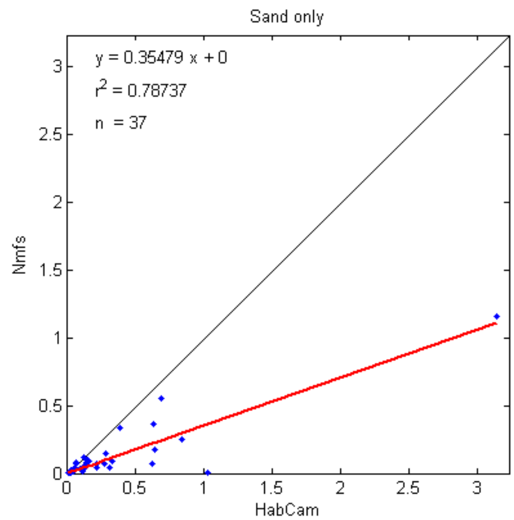
Appendix B9-Figure 25. Georges Bank and track lines of the R/V Hugh Sharp Leg 3 of the NMFS scallop survey (orange) and the regions where HabCam was deployed and collecting images (purple). Red dots and yellow stars are yellowtail flounder sightings

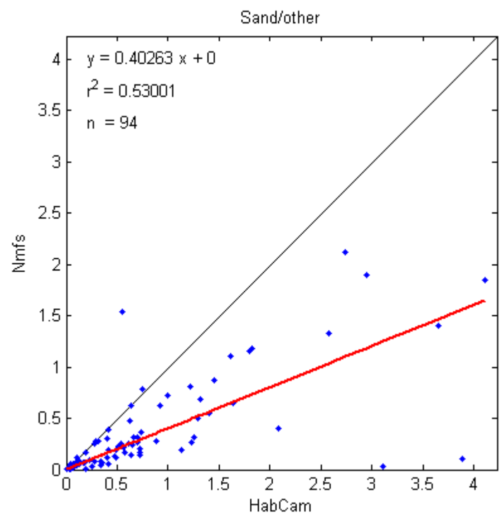


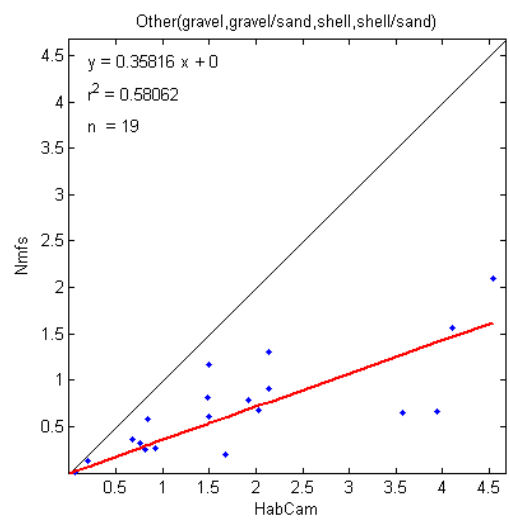
Appendix B9-Figure 26. Composite of example images of yellowtail flounder from Georges Bank.



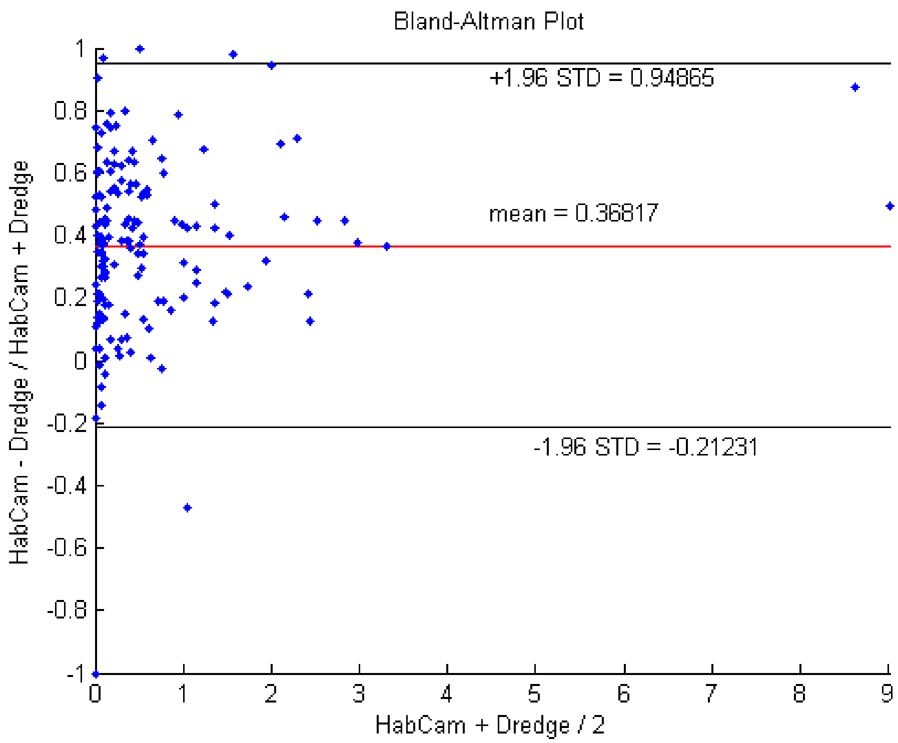








Appendix B9-Figure 27. Regressions of dredge survey estimates against HabCam estimates of scallop densities for both Years 2008 and 2009. Each point represents a single 1nm tow in the Georges Bank (a) or Mid Atlantic Bight (b) areas. Data broken out by substrate type in sand (c), sand plus shell hash (d), and gravel (e). The one to one correspondence line is plotted in black.



Appendix B9-Figure 28. Bland-Altman plot of the residuals for all joint tows in 2008 and 2009. The y axis represents (Dredge-HabCam)/(Dredge+HabCam) and the x axis is the mean of the two observations. The mean difference and the limits of agreement are also plotted.

<u>Assessment of Real and Potential Errors Associated with HabCam Image Data</u>

The sources of error to be assessed in this section are:

a) Border rules for measuring and counting scallops
b) Engineering Error
- Calibration of Field of View (FOV), complete camera model for intrinsic parameters, estimation of in-water, focal length, principle point, and pixel error
- Incorporation of extrinsic parameters for each image into calculation of FOV (area swept)- roll, pitch, heading, altitude

c) Human Error
- Analysis of measurement error both between individuals and within individuals using Intra Class Correlation

d) Imaging Error
- Scallop shells not orthogonal to camera axis

e) Total Measurement Error
- Analysis of shell height measurement error relative to NMFS dredge survey in NLSCA

f) Errors in interpolation of 1D data into 2D

- Kriging correlograms and variograms
- Variance within and between gridded cells
- Non-model based assessment of biomass

Each of these sources of potential error is discussed below.

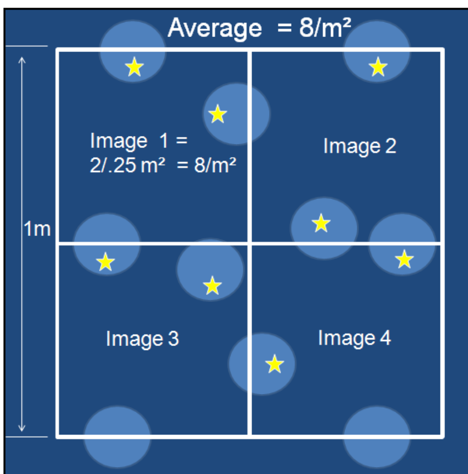
HabCam border rules

The purpose of the HabCam rules for border effects is to reduce undercounting or over counting due to animals being on the edge of images. The desired outcome is to count scallops on the edge of images exactly half the time. To achieve this, the following rules, which result in counting only scallops that have their centroid in the image, are followed (Fig. 29):

Primary Rule: Count all organisms that are more than half way in the image.

Secondary Rule: If the organism is exactly half way in the image, count only the organisms that are half way in the top and right sides. This process is identical to that described for counting blood cells on a Spears-Levy Hemacytometer and eliminates the need for altering the field of view (FOV) on an image to account for image basis.

a



Appendix B9-Figure 29. a) Four cases (represented as white squares) are shown in a 1 m^2 area to demonstrate HabCam border rules. Circles represent scallops. Stars represent scallops which would be counted in each of the 4 images. b) A HabCam image with scallops labeled “Yes” or “No” designating whether they would be measured using border rules.

b

Engineering errors-Calibration of Field of View (FOV)

Calibration of an optical system must include a complete camera model for intrinsic parameters, estimation of in-water focal length, principle point, and pixel error, followed by image correction by employing extrinsic parameters collected for each image.

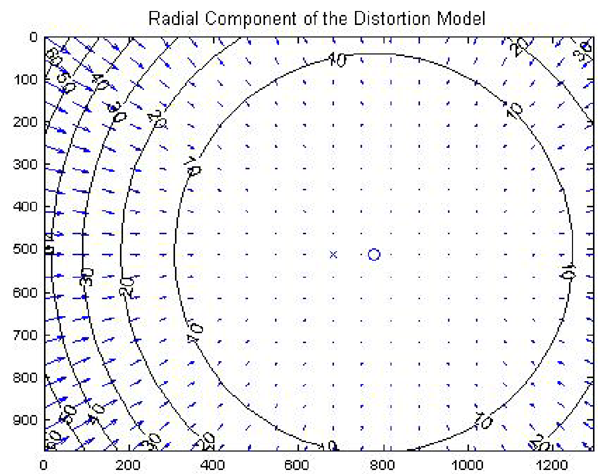
The intrinsic parameters for the HabCam camera were calculated using images of a 1m^2 target marked off at 10cm intervals in a 4 m deep seawater tank. The HabCam vehicle was positioned above the target at various altitudes (1-3m), roll, and pitch (0 and 20 degrees). Twenty eight images representing a range of positions were used for calibration of the camera with the Calibration Toolbox in Matlab.

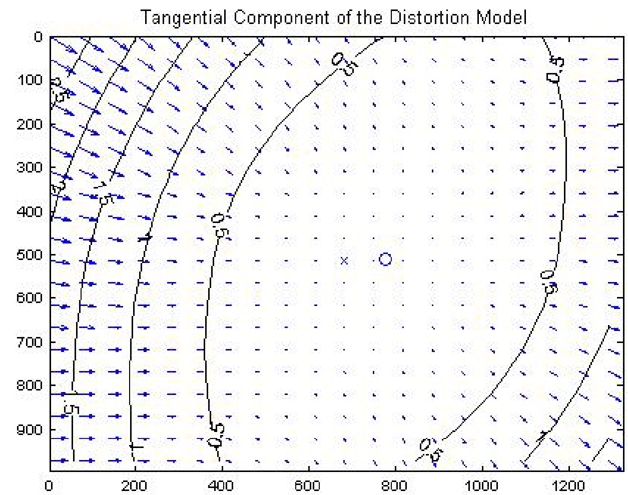
Engineering errors-Intrinsic parameters
(based on 28 images of target at different altitudes and orientations)

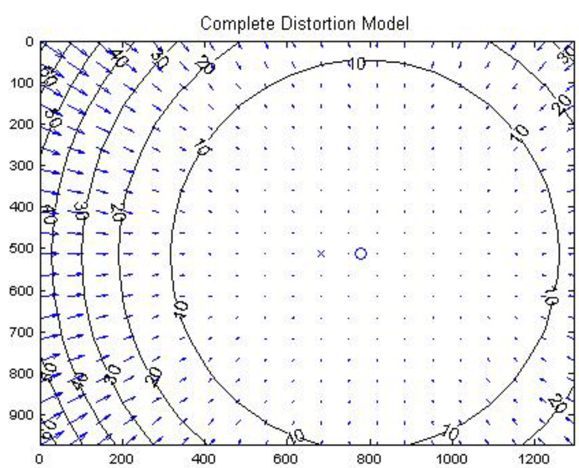
Focal Length: fc = [2773.25504 2764.28859] ± [7.18117 7.13362]
Principal point: cc = [778.19667 509.00401] ± [4.13012 3.80811]
Skew: alpha_c = [0.00000] ± [0.00000] => angle of pixel axes = 90.00000 ± 0.00000 degrees
Distortion: kc = [-0.31591 0.14388 0.00070 0.00138 0.00000] ± [0.00702 0.02649 0.00038 0.00056 0.00000]
Pixel error: err = [0.53035 0.50489]

The numerical errors are approximately three times the standard deviations
Intrinsic pixel error = +/- 1.59 pixels
Resolution range f (FOV): 0.37 – 0.89 mm/pixel
Intrinsic real-world error : 0.58 – 1.41 mm

These values provide error bounds on the resolution and accuracy of the camera system in water. Plots of the relative errors show that the camera CCD chip, lens and housing window are slightly out of alignment in both radial and tangential attitudes (Fig. 30). The pixel resolution is a function of FOV, which in turn is a function of altitude off the bottom. In calibrated screen measurement space, the overall measurement error is between 0.58 and 1.41 mm.







Appendix B9-Figure 30. Radial, tangential, and complete distortion model for the HabCam camera.

Distortion in each image is first corrected using the intrinsic parameters given above (Fig. 31).
KK = [fc(1) alpha_c*fc(1) cc(1);0 fc(2) cc(2) ; 0 0 1];
where the KK matrix is the uncorrected image matrix.

r2_extreme = (nx^2/(4*fc(1)^2) + ny^2/(4*fc(2)^2));

dist_amount = 1; %(1+kc(1)*r2_extreme + kc(2)*r2_extreme^2);

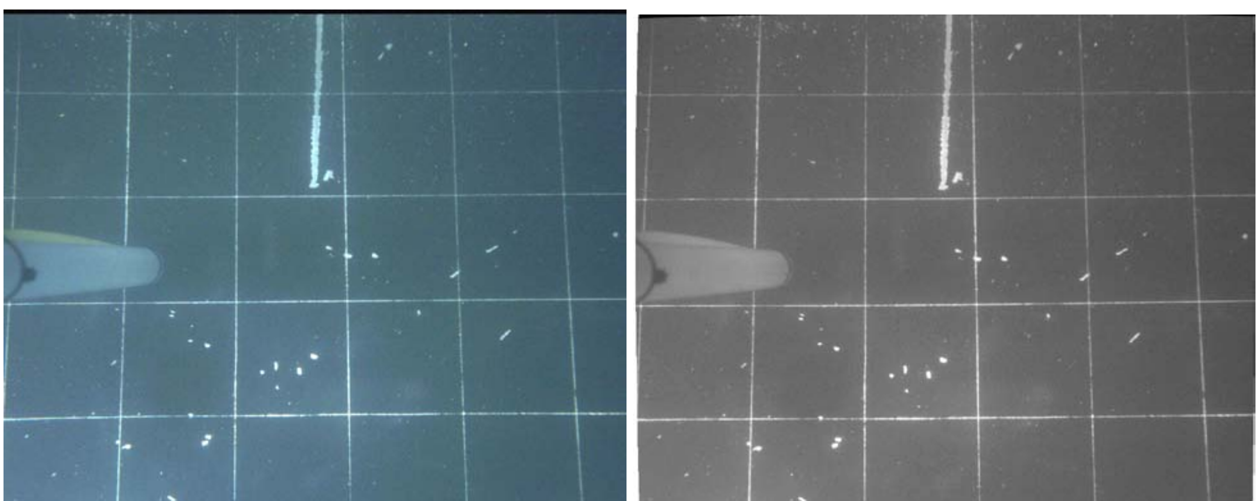
fc_new = dist_amount * fc;

KK_new = [fc_new(1) alpha_c*fc_new(1) cc(1);0 fc_new(2) cc(2) ; 0 0 1];

KK_new is the corrected image matrix.

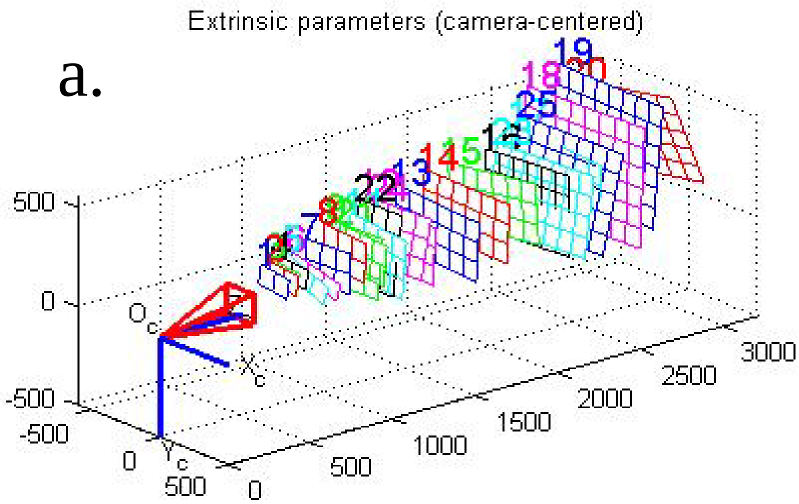
[I2] = rect(I,eye(3),fc,cc,kc,KK_new);

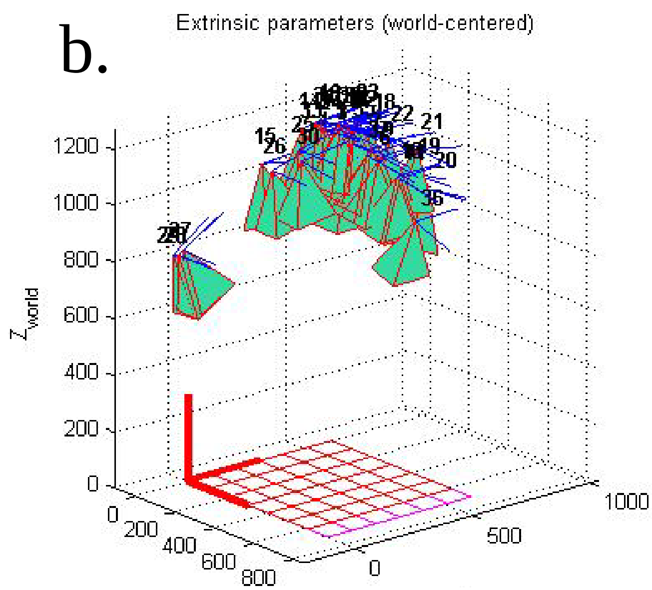
Where I is the distorted image and I2 is the undistorted image



Appendix B9-Figure 31. Correction of a distorted image of the calibration target in water (left) using intrinsic camera parameters. The corrected image (right) shows straight rather than curved lines particularly towards the corners of the image.

Extrinsic parameters relate to the combination of intrinsic parameters plus the orientation of the camera relative to the image plane. The calibration matrix is built up from the 28 views indicated in Figure 32.





Appendix B9-Figure 32.

a) Camera centric views of 28 orientations and altitudes to build extrinsic parameter list.

b) World centric views of 28 orientations and altitudes.

Engineering errors-Calculation of extrinsic parameters

Cross over points in the calibration chart are automatically detected and their locations in pixel space extracted before calculation of extrinsic parameters (Fig. 33).

The extrinsic parameters are encoded in the form of a rotation matrix (**Rc_ext**) and a translation vector (**Tc_ext**). The rotation vector **omc_ext** is related to the rotation matrix (**Rc_ext**) through the Rodrigues formula: **Rc_ext = rodrigues(omc_ext)**.

Let **P** be a point space of coordinate vector **XX = [X;Y;Z]** in the grid reference frame **(O,X,Y,Z).**

Let $\mathbf{XX_c = [X_c;Y_c;Z_c]}$ be the coordinate vector of **P** in the camera reference frame $\mathbf{(O_c,X_c,Y_c,Z_c)}$.
Then **XX** and $\mathbf{XX_c}$ are related to each other through the following rigid motion equation:
$\mathbf{XX_c = Rc\_ext * XX + Tc\_ext}$

In addition to the rigid motion transformation parameters, the coordinates of the grid points in the grid reference frame are also stored in the matrix **X_ext**.

Each image taken by HabCam has its own unique set of extrinsic parameters.

Extrinsic parameters for an example image:

Translation vector:
Tc_ext = [-225.840216 -130.369514 608.628548]

Rotation vector:
omc_ext = [-2.148393 -2.284790 -0.123388]

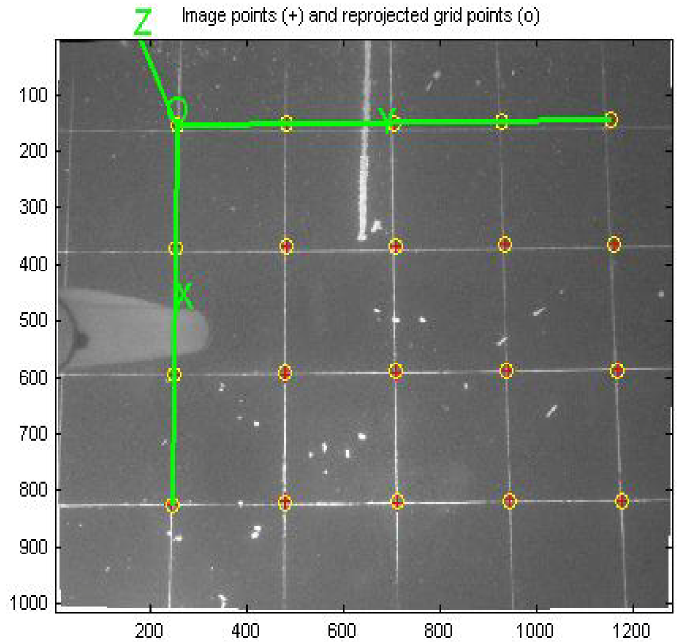
Rotation matrix:

Rc_ext = [-0.062925	0.996680	0.051672
0.996448	0.059838	0.059254
0.055966	0.055217	-0.996905]

Reprojection Pixel Error: err = [2.00116 1.26492]

Extrinsic pixel error = +/- 2 pixels
Resolution range (FOV): 0.37 – 0.89 mm/pixel

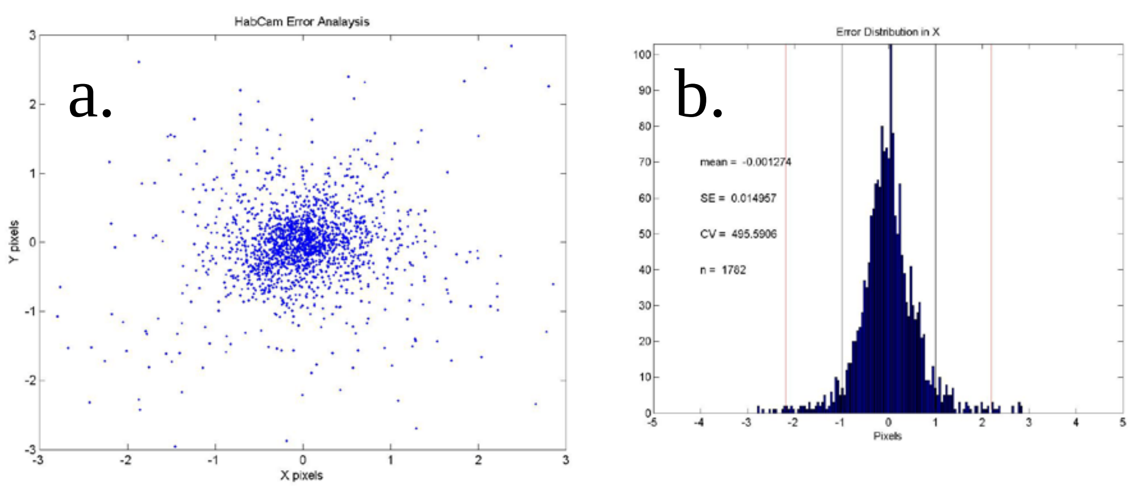
The extrinsic real-world error becomes: 1.11 – 1.78 mm (under best optical conditions)



Pixel error	= [1.482, 1.706]	
Focal Length	= (2731.98, 2749.06)	+/- [231.9, 234.3]
Principal Point	= (687.941, 743.467)	+/- [22.87, 57.85]
Skew	= 0	+/- 0
Radial coefficients	= (0.0541, 1.188, 0)	+/- [0.08834, 0.9791, 0]
Tangential coefficients	= (0.03035, -0.002405)	+/- [0.007949, 0.005481]

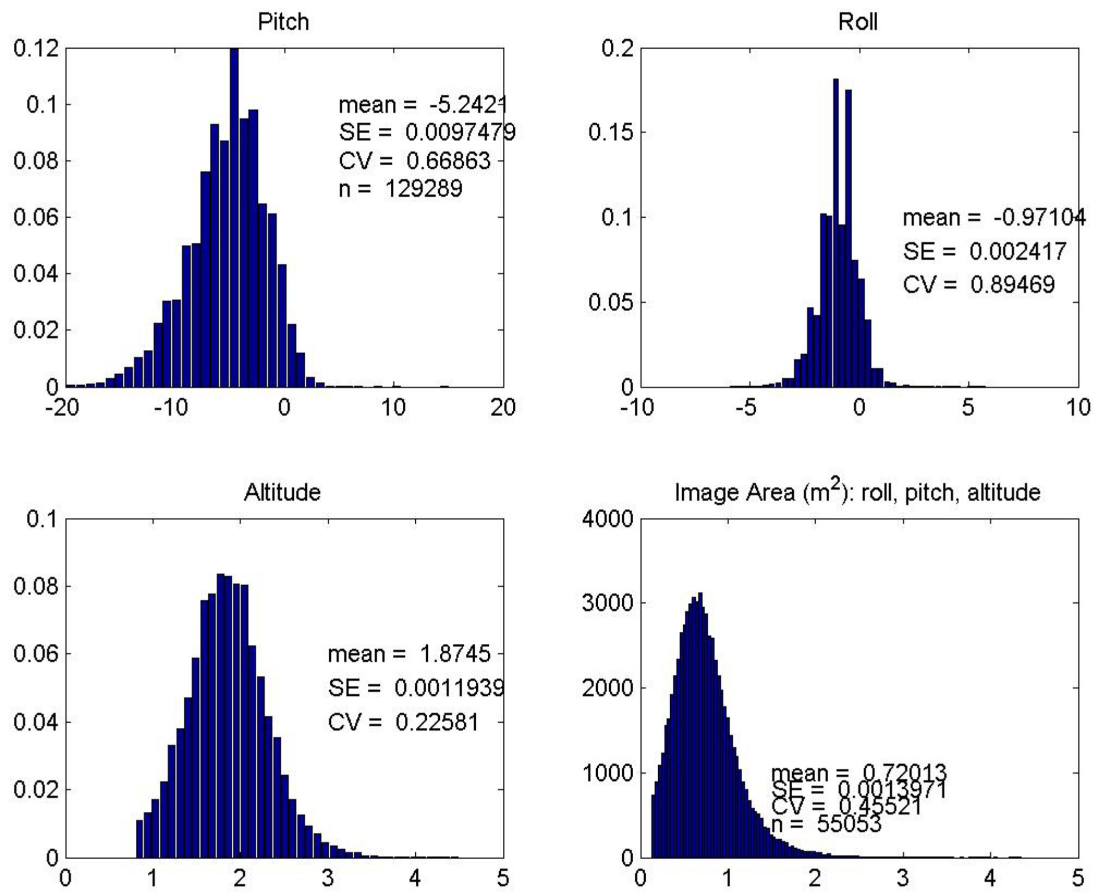
Appendix B9-Figure 33. Extrinsic parameters for image 24 above, as an example.

Pixel error in X and Y can be visualized as a scatter plot and frequency distribution (Fig. 34). Note that 99% of values are less than 2.2 pixels.

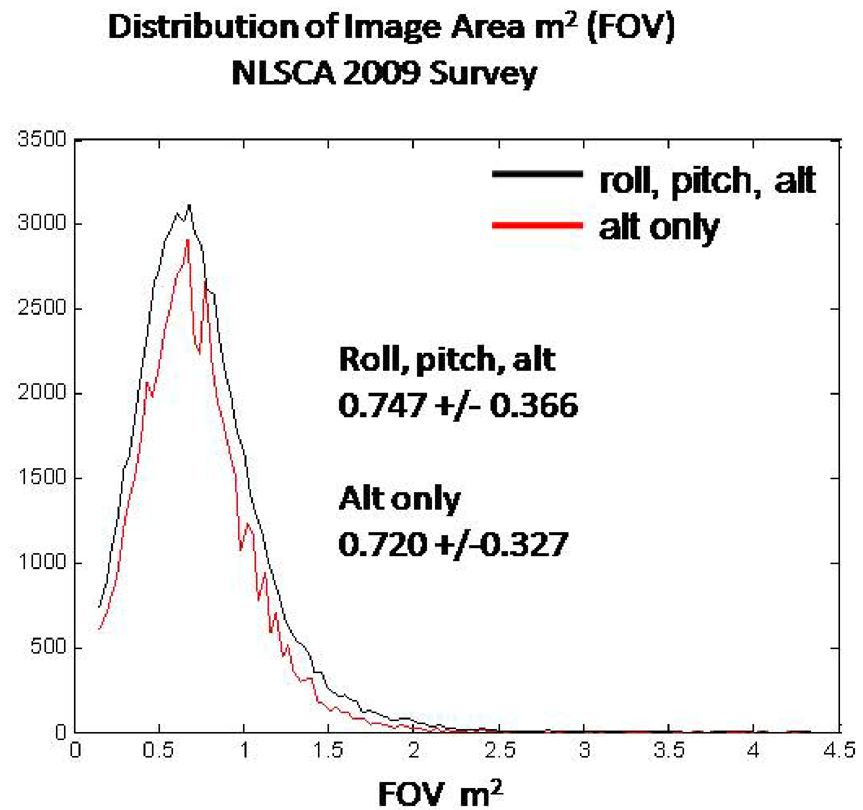


Appendix B9-Figure 34. (a) Scatter plot of pixel error around the origin. (b) Frequency distribution of pixel error along x axis.

Frequency distributions of pitch, roll, altitude, and image area (FOV) for the NLSCA 2009 survey based on 129,289 images are shown in Figure 35. The mean pitch was -5.24 degrees indicating that, on average, the nose of the vehicle pointed down slightly. Downward pitch is part of the system design and tends to stabilize the vehicle while underway. Mean roll was -0.97 with very little variation indicating the vehicle is quite stable, laterally. Altitude measurement varied from <1 to 4.5 m off the bottom with a mean of 1.87m. Images below 1 m were out of focus and removed from the image database. Images taken higher than 3 m were typically not sufficiently clear, due to turbidity, to be useful and were also not used. Taking roll and pitch into account using the extrinsic equations present above, the FOV ranged from 0.2 to >4m^2 with a mean of 0.72 m^2 . 95% of the calculations for FOV fell between 0.4 and 1.5 m^2. Figure 36 shows a comparison between FOV calculated with and without the use of roll and pitch, i.e., directly from the altitude, only. Incorporation of roll and pitch into the geometric projection of the FOV has an effect of broadening and smoothing the frequency distribution of values without changing the mean.



Appendix B9-Figure 35. Frequency distributions of pitch, roll, altitude, and image area (FOV) for the NLSCA 2009 survey.

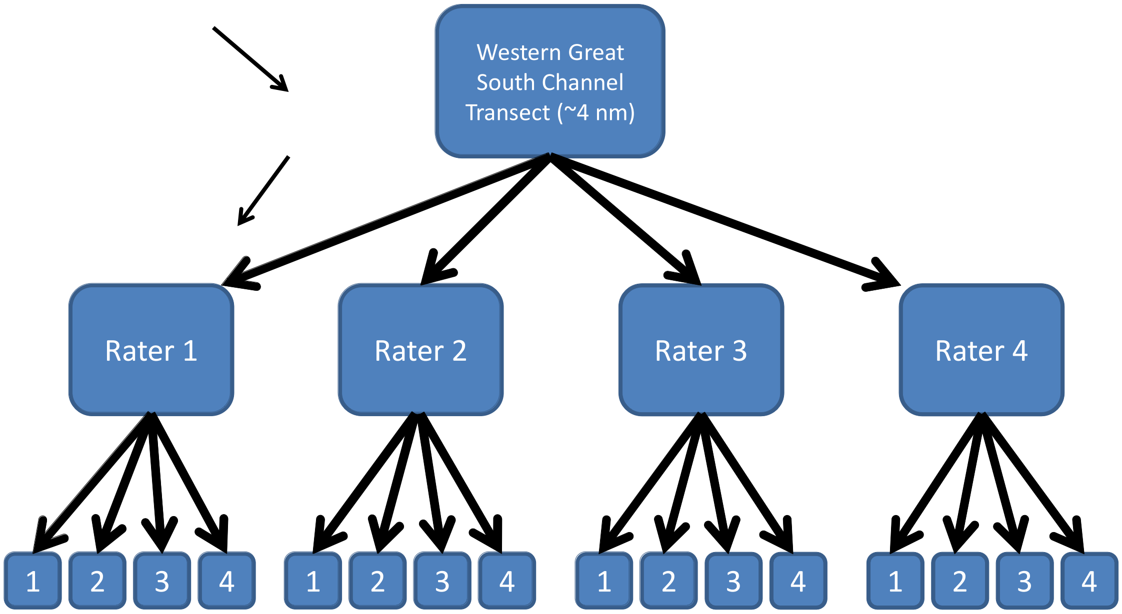


Appendix B9-Figure 36. Comparison between calculations of FOV with and without the effect of roll and pitch on geometric projection of FOV. Note broadening and smoothing of the distribution without a significant change in the mean, when projected geometry is used in conjunction with altitude.

Human errors: Analysis of measurement error both between individuals and within individuals
It was desired to estimate the level of error associated with the manual screen measurement of scallops both within a given technician and between technicians. The former would provide insight into measurement repeatability and the latter into systematic bias between individuals.

To accomplish this, we assigned four identical 4.2 nm long image transects containing 4,432 images from Western Great South Channel to six individuals (raters) (Fig. 37). Raters measured scallops using MIP under the same measurement rules as would be used under normal conditions (edge effects, height vs. width, etc).

ICC Intra Class Correlation Analysis



Two-way mixed effects model

Xij = u + ri + ci = rcij + e ij

u: population mean, r: row effects, c: column effects, e: residual effects

281 scallops x 4 raters x 4 passes = 4,496 measurements

Appendix B8-Figure 37. Inter Class Correlation analysis of scallop shell height measurements. Four individuals measured scallops from one transect four times.

Appendix B9-Table 1. Summary statistics in pixels. N = 277 for each run. A total of 4,432 scallops were measured. KLB, ADY, PK, and DPF are initials of the four raters.

rater	**KLB**				
	run1	**run2**	**run3**	**run4**	**mean**
mean	132.41	132.21	132.58	132.28	132
STD	31.61	31.61	31.44	31.69	
SE	1.89	1.91	1.88	1.9	
rater	**ADY**				
	run1	run2	run3	run4	mean
mean	128.48	128.46	128.41	128.75	128
STD	31.42	31.55	31.44	31.78	
SE	1.88	1.89	1.88	1.9	
rater	**PK**				
	run1	run2	run3	run4	mean
mean	135.57	135.88	134.95	134.28	135
STD	31.86	31.94	31.81	31.63	
SE	1.91	1.91	1.91	1.9	
rater	**DPF**				
	run1	run2	run3	run4	mean
mean	128.36	127.39	127.48	127.56	127
STD	31.74	31.63	31.45	31.57	
SE	1.9	1.9	1.89	1.89	

In most cases, mean within rater measurements were either accurate to the same number of pixels or within one pixel suggesting that within rater variability was extremely low (Table 1). Between rater variability was greater than within rater variability with mean values of 132, 128, 135, 127, providing a range of 135 to 127, or 8 pixels. Given the resolution range for varying FOV presented above (i.e., 0.37 – 0.89 mm/pixel), an error of 8 pixels represents a real-world error of 3.0 to 7.1 mm.

Inter and intra-Class correlations were analyzed using ICC, Intra Class Correlation analysis (McGraw and Wong, 1996). A two-way mixed effect model
Xij = u + ri + ci = rcij + eij
u: population mean, r: row effects, c: column effects, e: residual effects

was used to test the hypotheses that there is no difference between scallop measurements made by the same rater four times, and that there is no difference between individual raters.

ICC Type C-1: Tests the degree of consistency among measurements

r = (MSR - MSE) / (MSR + (k-1)*MSE);
F = (MSR/MSE) * (1-r0)/(1+(k-1)*r0);
df1 = n - 1;
df2 = (n-1)*(k-1);
p = 1-fcdf(F, df1, df2);

ANOVA Table

Source	SS	df	MS	F	Prob>F
Time	143.5	3	47.8	6.28	0.0003
Group	40170.4	3	13390.1	3.36	0.0182
Ineratcion	480.6	9	53.4	7.01	0
Subjects (matching)	4398738.9	1104	3984.4	522.79	0
Error	25241.8	3312	7.6		
Total	4464775.2	4431			

r = 0.9842
LB = 0.9827
UB = 0.9857
p = 0

ICC Type A-1: Test the degree of absolute agreement among measurements.

r = (MSR - MSE) / (MSR + (k-1)*MSE + k*(MSC-MSE)/n);

a = (k*r0) / (n*(1-r0));
b = 1 + (k*r0*(n-1))/(n*(1-r0));
F = MSR / (a*MSC + b*MSE);
df1 = n - 1;
df2 = (a*MSC + b*MSE)^2/((a*MSC)^2/(k-1) + (b*MSE)^2/((n-1)*(k-1)));
p = 1-fcdf(F, df1, df2);

ANOVA Table

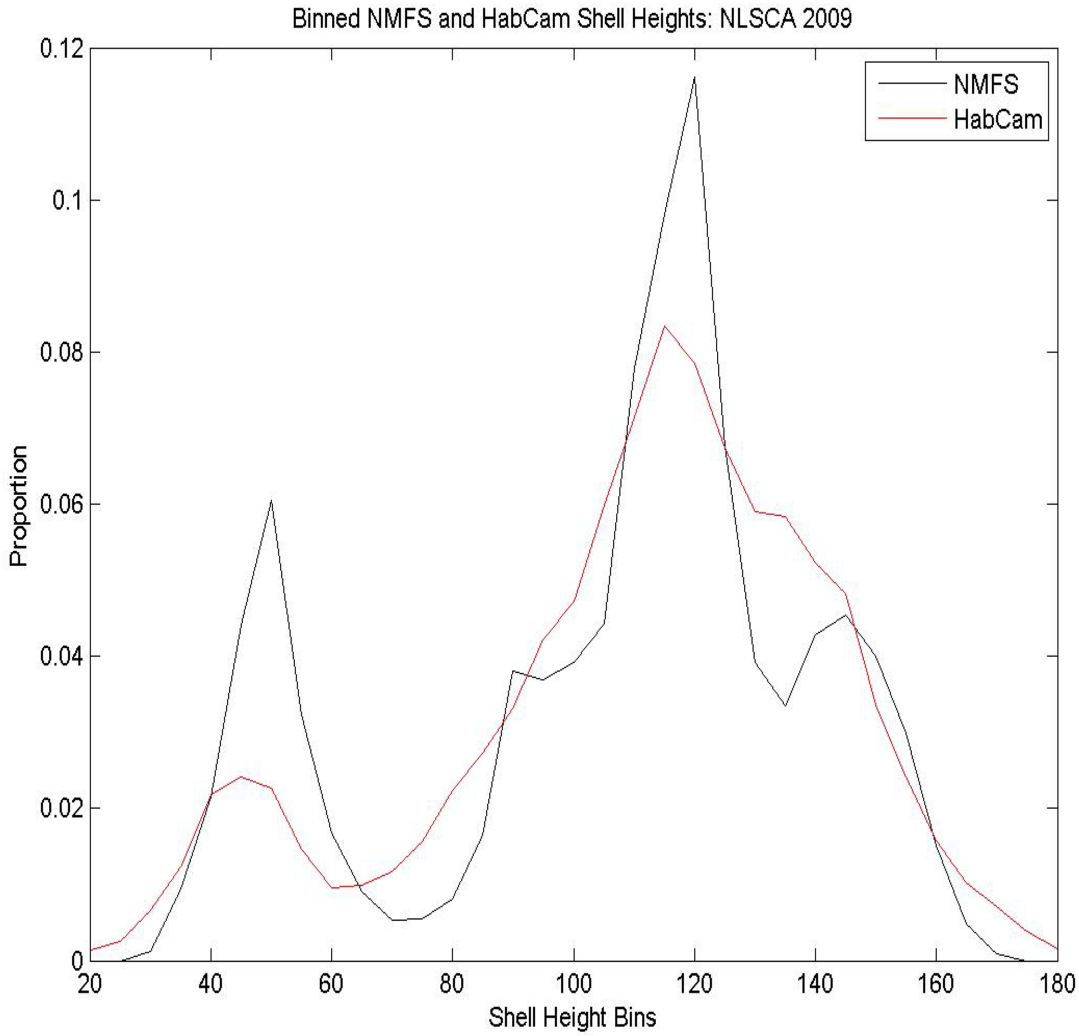
Source	SS	df	MS	F	Prob>F
Time	143.5	3	47.8	6.28	0.0003
Group	40170.4	3	13390.1	3.36	0.0182
Ineratcion	480.6	9	53.4	7.01	0
Subjects (matching)	4398738.9	1104	3984.4	522.79	0
Error	25241.8	3312	7.6		
Total	4464775.2	4431			

r = 0.9796
LB = 0.9695
UB = 0.9856
p = 0

Summary
There is no difference in measurements made by the same individual raters or between individual raters.

Human errors-Analysis of shell height measurement error
Shell height measurements from HabCam images taken during the 2009 Nantucket Lightship survey were compared with shell height measurements made from 12 dredge tows on 2009 Leg 2 of the R/V Hugh Sharp during normal survey operations. The HabCam survey was conducted in early June 2009 while the NMFS survey was conducted in min July, 2009.

Frequency distribution for shell height measurements for HabCam and NMFS dredge survey show surprising similarity in overall pattern (Fig. 38). Since the NMFS survey was conducted about five weeks following the HabCam survey, the shift in mode of the NMFS data for small scallops can be accounted for by growth. The tails of the distribution for HabCam data are spread out more than for the NMFS data suggesting a source of measurement error. There is no indication of selectivity by either sampling approach.



Appendix B9-Figure 38. Shell height size frequency distributions for NMFS (black) and HabCam (red) measurements in the NLSCA. HabCam surveyed in early June while NMFS surveyed in mid July. Note a shift to the right of the mode for small scallops in NMFS data relative to that for HabCam probably due to growth.

Shell heights made from HabCam and NMFS dredge survey were analyzed sing the approach described by Jacobson et al. 2010). Accuracy (RMSE, root mean square error) , bias (HabCam-Dredge), and precision (STD) were calculated.
HabCam measurements were positively biased relative to NMFS data by 3.7%. Percentage square root of the mean square error was 3.70%. Both NMFS and HabCam distributions were negatively skewed and more peaked relative to normal distributions.

stat		NMFS		HabCam
n		4,178		13,576
bias		NA		3.8
min		30		20
max		170		180
avg		106.4		110.3
%bias		NA		3.70%
STD		33.4		32.2
CV		31.4		27.90%
RMSE		NA		3.8
% RMSE		NA		3.70%
skewness (g1)		-0.59		-0.66
kurtosis (g2)		2.41		2.98

Summary of error analysis

Measurement error can come from a number of sources including intrinsic error in camera calibration, extrinsic error due to camera orientation and altitude relative to geometric projection on the image plane, and errors associated with human operators measuring scallops on the computer screen. Given the resolution range for varying FOV presented above (i.e., 0.37 – 0.89 mm/pixel), a real world for each source may identified.

Source	**+/- mm error**
Intrinsic	0.58 – 1.41
Extrinsic	1.11 – 1.78
Within operator	0.5
Between operator	3.0 – 7.1

Clearly, the magnitude of between operator errors dominates the overall potential for measurement error. However, once this source of variability is identified and characterized for each individual technician measuring scallops, a correction factor could be applied for each individual to normalize the results. In addition, automated counting and sizing of scallops and other targets is improving and will eventually be used in conjunction with manual measurements to produce more accurate results.

Appendix B10: Estimation of survey dredge efficiency relative to HabCam.

Tim Miller and Dvora Hart, Northeast Fisheries Science Center, Woods Hole, MA.

Introduction

Using data from a paired-tow calibration experiment, the goal is to estimate the efficiency of the NMFS scallop survey dredge relative to that of the HabCam. The HabCam survey instrument is usually assumed to be 100% efficient so that the absolute efficiency of the survey dredge can be estimated. However, the relative efficiency of the NMFS survey dredge can be estimated without this assumption.

Methods

The data we have to work are for both HabCam and survey dredge at over 140 stations. For the HabCam, we have a number of images of the substrate along a track at each station. For each image, we have the numbers of scallops as well as the estimated area covered by the image. The HabCam captures images continuously along each track, but a thinned subset are used in our analyses. Thinning is intended to make serial correlation of the images within a station negligible. For the dredge, we have the total number of scallops captured at each station as well as an estimate of the swept area.

Statistical models

For these analyses, we consider different probability models for the HabCam and dredge data, but common to all models is our assumption that the expected catch in numbers of the dredge at station i is

$$E(N_{Di} \mid \delta_{Di}, A_{Di}) = q_D \delta_{Di} A_{Di} \qquad (1)$$

and that of the HabCam for photo j at station i is

$$E(N_{Hij} \mid \delta_{Hij}, A_{Hij}) = q_H \delta_{Hij} A_{Hij} \qquad (2)$$

where δ_{Di} and δ_{Hij} are the average density available to the dredge over the entire tow and the average density in the HabCam for image j at station i, and q_D and q_H are the catchabilities for the dredge and HabCam. The respective areas swept by the dredge and in the image j from the HabCam are A_{Di} and A_{Hij} which are assumed known.

The simplest probability model for count data is the Poisson distribution and in gear comparison studies it is common to make use of binomial models which are conditional on the total catch at a given station (e.g., Millar 1992, Lewy et al. 2004). If the density was constant across all of the HabCam images and the dredge, the binomial model would be useful for these data (Appendix B9). However, densities may vary within a station and the numerous HabCam observations at each station allow us to investigate the plausibility of this assumption.

Suppose that each datum for the HabCam and the dredge arises from a Poisson distribution with mean (and variance) given by eqs. 1 and 2, respectively. If we assume the densities for the HabCam photos at station i to be independently and identically distributed as

$$\delta_{Hij} \sim \text{Gamma}\left(\Delta_i, \tau_{1i}\right)$$

where $E(\delta_{Hij}) = \Delta_i$ is the mean density and the variance is $V(\delta_{Hij}) = \Delta_i^2 / \tau_{1i}$, then the catches in each photo arise marginally from a negative binomial distribution with mean and variance

$$E(N_{Hij} \mid A_{Hij}) = q_H \Delta_i A_{Hij} = \mu_{Hi} A_{Hij}$$

$$V(N_{Hij} \mid A_{Hij}) = E(N_{Hij}) + E(N_{Hij})^2 / \tau_{1i}$$

where $\mu_{Hi} = q_H \Delta_i$. As the dispersion parameter τ_{1i} increases, the variability in densities within a station decreases and the observed number in the image approaches the Poisson in distribution.

We can also model variability in densities for the dredge,

$$\delta_{Di} \sim \text{Gamma}\left(\Delta_i, \tau_{2i}\right)$$

so that the marginal distribution of the number caught in the dredge is negative binomial with

$$E(N_{Di}) = q_D \Delta_i A_{Di} = \rho \mu_{Hi} A_{Di}$$

and

$$V(N_{Di}) = E(N_{Di}) + E(N_{Di})^2 / \tau_{2i}\,.$$

The dispersion parameter for the dredge is distinguishable from that of the HabCam data which allows the variability among the observed average densities in HabCam images to differ from that of the dredge. This model is estimable when there is only a single observation from the dredge at each station because the mean is related to that of the HabCam images by the relative catchability parameter, ρ, which is informed by data from all stations, and because the mean catch per unit area of HabCam images μ_{Hi} is informed by all of the HabCam images at the station. Therefore, the single observation by the dredge can inform the dispersion parameter τ_{2i}. Note that simpler models where $\Delta_i = \Delta$, $\tau_{1i} = \tau_1$, or $\tau_{2i} = \tau_2$ are special cases.

The relative efficiency of the dredge to the HabCam may differ by substrate type. We observe the substrate in each HabCam image, but the dredge track may cover various substrates which are not directly observed. The lack of these observations for the dredge makes estimation of relative efficiency for specific substrates impossible, but because certain substrates are known to be more prevalent in particular strata, we may consider using these broader regions as proxies that can be used as covariates. As such, we defined three regional indicators for the stations in this study depending on the strata where they occur. Sandy bottom is predominant in the Mid-Atlantic region which includes strata 6130, 6140, 6150, 6180, and 6190 and Georges Bank strata 6460, 6470, 6530, 6540, 6550, 6610, 6621, and 6670 whereas rock and gravel substrates are common in Georges Bank strata 6490, 6500, 6510, 6520, 6651, 6652, 6661, 6662, and 6710. We also formed an alternative set of two regional indicators where the two regions with predominantly sandy bottom were combined.

Model fitting

We fit models using programs in AD Model Builder (ADMB 2009). The likelihood function depends on the assumptions about the parameters and distributions and the parameters were estimated in log-space to avoid boundary conditions.

We restricted the data used for model fitting to stations where there was more than 1 scallop observed in the HabCam images because estimating a positive mean catch per image area at the station is impossible when no scallops are observed. We also removed data for stations where there were less than 2 non-zero counts on HabCam images because fitting negative binomial models for these data at each station requires a sufficient number of positive observations to provide estimates of uncertainty. Ultimately, we used data from 140 of the 146 stations in the original data set.

During the analyses, we discovered that fitted models where the negative binomial assumption was made at all stations for the HabCam data converged in the parameter space where the Hessian matrix was not positive definite. Upon inspection, several of the station-specific dispersion parameters were estimated at extremely large values which implied that the data at these stations were better treated with a Poisson model. We fit both negative binomial and Poisson models to the HabCam data at each station and compared the fits by AIC_c (Burnham and Andersen 2002) to determine which stations we could assume were Poisson distributed. These results were corroborated by inspection of the magnitude of the estimated quasi-likelihood dispersion parameters and negative binomial dispersion parameters at each station.

The full set of models that we fit to estimate relative efficiency of the dredge is provided in Table 1. In the first, most basic, set of models (P/P), we assume the Poisson distribution for all of the data for the HabCam and the dredge. In the second set of models (P/NBP), the dredge data are Poisson distributed and the HabCam data from each station arise from either a Poisson or negative binomial distribution depending on the AIC_c values of those models at each station. For the third set of models (NBP/NBP), both the dredge and HabCam data at each station are either Poisson or negative binomial distributed based on the AIC_c values of the model fits to the HabCam data. In the last set of models (NB/NBP), all of the dredge data are negative binomial distributed.

Within each set of models we allow different parameterization assumptions for specific models (Table 1). The marginal scallop density at a given station may either be constant or station-specific. The relative efficiency may either be constant, region-specific (substrate proxy), or station-specific. For models with negative binomial assumptions, dispersion parameters for the HabCam data may either be constant or station-specific.

One last model in the NB/NBP set was fit where the negative binomial dispersion parameter for the dredge was allowed to be station-specific, but similar to the HabCam data, there were stations where the dispersion parameter was estimated extremely high and variance estimation was not possible. We assigned Poisson distributions to stations where the dispersion parameter estimates were greater than 1000.

Results

As one would expect, the use of AIC_c to determine whether the Poisson is preferred by station corresponds well to the magnitude of the estimated quasi-likelihood dispersion parameter for the corresponding stations (Figure 1). When the quasi-likelihood dispersion parameter is equal to one, the variance is equal to the mean which is an implicit assumption for the Poisson model. Because the variance is always greater than the mean for the negative binomial model, the Poisson model which is more parsimonious is expected to have a lower AIC_c value if the quasi-likelihood dispersion parameter is approximately equal or less than one. The AIC_c criterion also corresponds well with magnitude of the estimated negative binomial dispersion parameter when that model is fitted (Figure 2). When the negative binomial dispersion parameter is large the data approach Poisson in distribution.

That the negative binomial assumption is better for many stations is also reflected in lower AIC_c values (over all stations) for fitted models that allow it (Table 2). The models where the Poisson distribution is assumed for both the dredge and HabCam observations at all stations had the poorest fits based on AIC_c. The lowest AIC_c value for any P/P model was approximately 10,000 units greater than the best fits among other classes of models that we considered.

Fits for two of the models converged but the Hessian was not positive definite and variance estimation was not possible (NBP/NBP M_5 and NB/NBP M_5). These models were among the best fits with regard to AIC_c, but a model with the Poisson assumption for the dredge data and negative binomial or Poisson assumptions for the HabCam data provided the same maximized log-likelihood with fewer parameters and a positive-definite hessian matrix (P/NBP M_5). Although P/NBP M_5 provided the best fit, it is parameterized with station-specific relative efficiencies which cannot be used to infer the efficiency of the dredge in previous years.

The model with the lowest AIC_c that can be used to infer efficiency of the dredge throughout the time series is NB/NBP M_6 which allowed different relative efficiencies for the regions predominant in gravel and sand. The estimated relative efficiency of the dredge is 0.462 (0.006 SE) in the sandy regions and 0.401 (0.011 SE) in the gravel regions.

Discussion

We found that among the fitted models the best fit was provided by allowing the calibration factors to be station-specific. This was not practical for the uses here in the scallop assessment, but these results imply that there is substantial heterogeneity in the relative efficiency of the dredge. A better model would allow a further hierarchy to describe the variation in the relative efficiency, which is an important avenue of analyses in the future.

Of the applicable models that we fit, the best model allowed different relative efficiencies for the regions with predominantly sandy and gravel substrates. The higher relative efficiency of the dredge in the sandy region is expected because the dredge is intended to operate optimally in finer substrates rather than coarse substrates such as gravel and rock.

Finally, it should be noted that these analyses were carried out with swept areas for the dredge based on nominal tow path estimates. Work carried out concurrent to this study suggests that the true tow path is about 4-10% more than those used here. An additional adjustment to our estimates of survey dredge sampling efficiency may be required in some applications.

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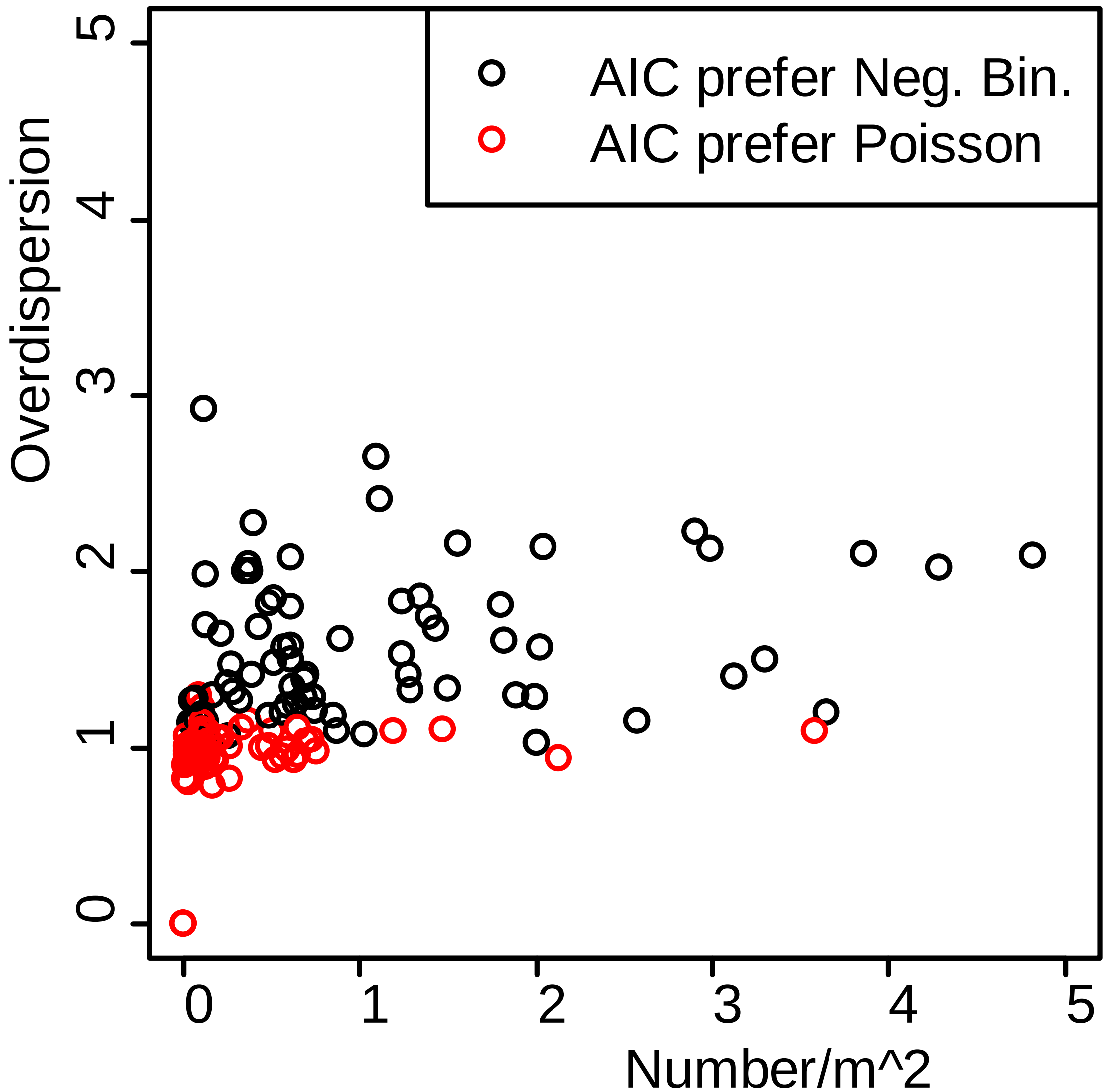
Appendix B10-Table 1. Models fitted to the HabCam and dredge data.

Model	Description	Parameters
P/P M_0	Dredge data and HabCam data are Poisson distributed. Density is constant, relative catchability is constant.	ρ and μ_H
P/P M_1	Dredge data and HabCam data are Poisson distributed. Density is station-specific, relative catchability is constant.	ρ and μ_{Hi}
P/P M_2	Dredge data and HabCam data are Poisson distributed. Density is station-specific, relative catchability is region-specific (Gravel/Sand).	ρ_s and μ_{Hi}
P/P M_3	Dredge data and HabCam data are Poisson distributed. Density is station-specific, relative catchability is region-specific (GB Gravel/GB Sand/MA Sand).	ρ_r and μ_{Hi}
P/P M_4	Dredge data and HabCam data are Poisson distributed. Density is station-specific, relative catchability is station-specific.	ρ_i and μ_{Hi}
P/NBP M_0	Dredge data are Poisson distributed and HabCam data are either Poisson or negative binomial distributed. Density is constant, relative catchability is constant, dispersion is constant	ρ , μ_H , τ_1
P/NBP M_1	Dredge data are Poisson distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is constant, dispersion is constant.	ρ , μ_{Hi} , and τ_1
P/NBP M_2	Dredge data are Poisson distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is constant, dispersion is station-specific.	ρ , μ_{Hi} , and τ_{1i}
P/NBP M_3	Dredge data are Poisson distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is region-specific (Gravel/Sand), dispersion is station-specific.	ρ_s , μ_{Hi} , and τ_{1i}
P/NBP M_4	Dredge data are Poisson distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is region-specific (GB Gravel/GB Sand/MA Sand), dispersion is station-specific.	ρ_r , μ_{Hi} , and τ_{1i}
P/NBP M_5	Dredge data are Poisson distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is station-specific, dispersion is station-specific.	ρ_i , μ_{Hi} , and τ_{1i}
NBP/NBP M_0	Dredge data and HabCam data at each station are either Poisson or negative binomial distributed. Density is constant, relative catchability is constant, dispersion parameters are constant.	ρ , μ_H , τ_1 , and τ_2
NBP/NBP M_1	Dredge data and HabCam data at each station are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is constant, dispersion parameters are constant.	ρ , μ_{Hi} , τ_1 , and τ_2
NBP/NBP M_2	Dredge data and HabCam data at each station are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is constant, HabCam dispersion is station-specific, dredge dispersion parameter is constant.	ρ , μ_{Hi} , τ_{1i} , and τ_2
NBP/NBP M_3	Dredge data and HabCam data at each station are either Poisson or negative binomial distributed with a common dispersion parameter. Density is station-specific, relative catchability is region-specific (Gravel/Sand), HabCam dispersion is station-specific, dredge dispersion parameter is constant.	ρ_s , μ_{Hi} , τ_{1i} , and τ_2
NBP/NBP M_4	Dredge data and HabCam data at each station are either Poisson or negative binomial distributed with a common dispersion parameter. Density is station-specific, relative catchability is region-specific (GB Gravel/GB Sand/MA Sand), HabCam dispersion is station-specific,	ρ_r , μ_{Hi} , τ_{1i} , and τ_2

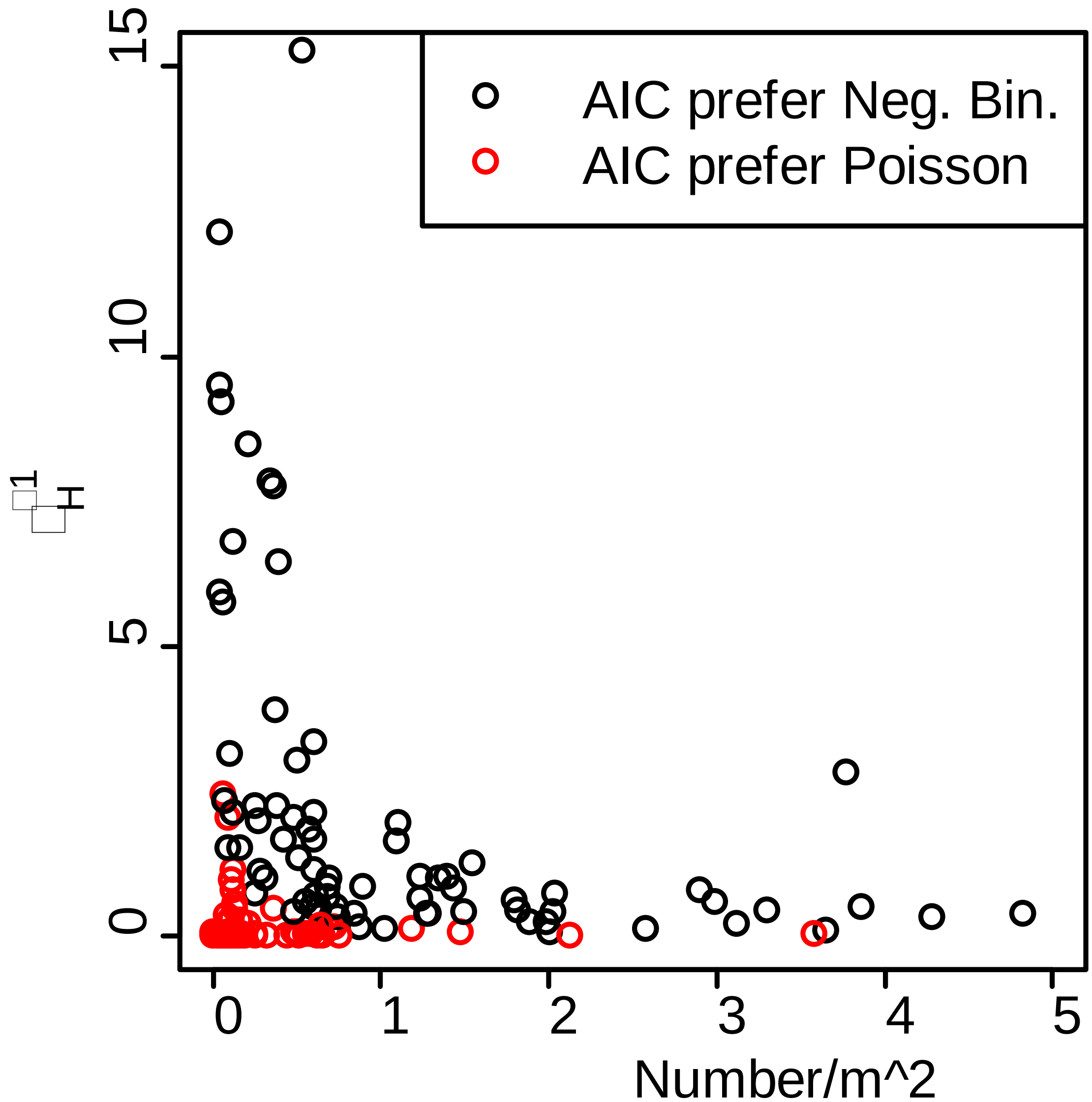
	dredge dispersion parameter is constant.	
NBP/NBP M_5	Dredge data and HabCam data at each station are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is station-specific, HabCam dispersion is station-specific, dredge dispersion parameter is constant.	ρ_i, μ_{Hi}, τ_{1i}, and τ_2
NB/NBP M_0	Dredge data are negative binomial distributed and HabCam data are either Poisson or negative binomial distributed. Density is constant, relative catchability is constant, dispersion parameters are constant	ρ, μ_H, τ_1, and τ_2
NB/NBP M_1	Dredge data are negative binomial distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is constant, dispersion parameters are constant.	ρ, μ_{Hi}, τ_1, and τ_2
NB/NBP M_2	Dredge data are negative binomial distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is constant, HabCam dispersion is station-specific, dredge dispersion parameter is constant.	ρ, μ_{Hi}, τ_{1i}, and τ_2
NB/NBP M_3	Dredge data are negative binomial distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is region-specific (Gravel/Sand), HabCam dispersion is station-specific, dredge dispersion parameter is constant.	ρ_s, μ_{Hi}, τ_{1i}, and τ_2
NB/NBP M_4	Dredge data are negative binomial distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is region-specific (GB Gravel/GB Sand/MA Sand), HabCam dispersion is station-specific, dredge dispersion parameter is constant.	ρ_r, μ_{Hi}, τ_{1i}, and τ_2
NB/NBP M_5	Dredge data are negative binomial distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is station-specific, HabCam dispersion is station-specific, dredge dispersion parameter is constant.	ρ_i, μ_{Hi}, τ_{1i}, and τ_2
NB/NBP M_6	Dredge data are either Poisson or negative binomial distributed and HabCam data are either Poisson or negative binomial distributed. Density is station-specific, relative catchability is region-specific (Gravel/Sand), HabCam dispersion is station-specific, dredge dispersion parameter is station-specific.	ρ_s, μ_{Hi}, τ_{1i}, and τ_{2i}

Appendix B10-Table 2. Number of parameters, maximized log-likelihood value and AIC_c for each fitted model. Log-likelihood and AIC_c values are in parentheses for models without invertible hessian matrices.

Model	No. Parameters	Log-Likelihood	AIC_c
P/P M_0	2	-278,850.0	557,704.0
P/P M_1	141	-72,019.1	144,320.8
P/P M_2	142	-71,581.8	143,448.2
P/P M_3	143	-71,578.9	143,444.4
P/P M_4	280	-62,693.0	125,948.5
P/NBP M_0	3	-250,341.0	500,688.0
P/NBP M_1	142	-63,288.6	126,861.8
P/NBP M_2	242	-60,667.5	121,820.8
P/NBP M_3	243	-60,511.4	121,510.7
P/NBP M_4	244	-60,503.0	121,495.9
P/NBP M_5	381	-57,444.1	115,654.8
NBP/NBP M_0	4	-94,524.7	189,057.4
NBP/NBP M_1	143	-58,743.3	117,773.2
NBP/NBP M_2	243	-57,932.3	116,352.5
NBP/NBP M_3	244	-57,924.1	116,338.1
NBP/NBP M_4	245	-57,918.5	116,328.9
NBP/NBP M_5	382	(-57,444.1)	(115,656.8)
NB/NBP M_0	4	-78,974.0	157,956.0
NB/NBP M_1	143	-58,706.5	117,699.6
NB/NBP M_2	243	-57,895.1	116,278.1
NB/NBP M_3	244	-57,893.8	116,277.5
NB/NBP M_4	245	-57,893.7	116,279.3
NB/NBP M_5	382	(-57,444.1)	(115,656.8)
NB/NBP M_6	315	-57,730.5	116,094.1



Appendix B10-Figure 1. Estimated overdispersion and mean observed number/m^2 from fitted quasi-likelihood model for HabCam count data at each station with log link. Red points indicate that the Poisson model was preferred based on AIC$_c$.



Appendix B10-Figure 2. Estimated (inverse) negative binomial dispersion parameter and mean observed number/m^2 for HabCam count data at each station. Red points indicate that the Poisson model was preferred based on AIC_c.

Appendix B11: Technical documentation for the CASA length structured stock assessment model.

Larry Jacobson, Northeast Fisheries Science Center, Woods Hole, MA.

[*This technical description is current through CASA version nc238 used for the SARC50 sea scallop assessment.*]

The stock assessment model described here is based on Sullivan et al.'s (1990) CASA model.[5] CASA is entirely length-based with population dynamic calculations in terms of the number of individuals in each length group during each year. Age is almost completely irrelevant in model calculations. Unlike many other length-based stock assessment approaches, CASA is a dynamic, non-equilibrium model based on a forward simulation approach. CASA incorporates a very wide range of data with parameter estimation based on maximum likelihood. CASA can incorporate prior information about parameters such as survey catchability and natural mortality in a quasi-Bayesian fashion and MCMC evaluations are practical. The implementation described here was programmed in AD-Model Builder (Otter Research Ltd.).[6]

Population dynamics

Time steps in the model are years, which are also used to tabulate catch and other data. Recruitment occurs at the beginning of each time step. All instantaneous rates in model calculations are annual (y^{-1}). The number of years in the model n_y is flexible and can be changed easily (e.g. for retrospective analyses) by making a single change to the input data file. Millimeters are used to measure body size (e.g. sea scallop shell heights). Length-weight relationships should generally convert millimeters to grams. Model input data include a scalar that is used to convert the units for length-weight parameters (e.g. grams) to the units of the biomass estimates and landings data (e.g. mt). The units for catch and biomass are usually metric tons.

The definition of length groups (or length "bins") is a key element in the CASA model and length-structured stock assessment modeling in general. Length bins are identified in CASA output by their lower bound and internally by their ordinal number. Calculations requiring information about length (e.g. length-weight) use the mid-length $_j$ of each bin. The user specifies the first length (L_{min}) and the size of length bins (L_{bin}). Based on these specifications, the model determines the number of length bins to be used in modeling as $n_L = 1 + \text{int}\left[(L_\infty - L_{\min})/L_{bin}\right]$, where L_∞ is maximum asymptotic size based on a von Bertalanffy growth curve supplied by the user, and int[x] is the integer part of x. The last length bin in the model is always a "plus-group" containing individuals L_∞ and larger. Specifications for length data used in tuning the model are separate (see below).

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[5] Original programming in AD-Model Builder by G. Scott Boomer and Patrick J. Sullivan (Cornell University), who bear no responsibility for errors in the current implementation.

[6] AD-Model Builder can be used to calculate variances for any estimated or calculated quantity in a stock assessment model, based on the Hessian matrix with "exact" derivatives and the delta method.

Growth

Growth is modeled in CASA using annual and/or monthly growth transition matrices supplied by the user. There are three options. Under option 1, the model ignores seasonal growth and calculated annual growth based on an annual growth transition matrix. Option 2 is similar but the annual growth matrix is constructed internally based on raw growth increments in the input file. Under option 3, monthly growth transition matrices from the input files are used in a variety of calculations (e.g. in tuning to body size composition data). Options 1 or 2 (annual growth only) are recommended at this time because of unresolved problems in using Option 3 with seasonal growth).

In population dynamics calculations, individuals in each size group grow (or not) at the beginning of the year, based on the annual growth transition matrix $\boldsymbol{P_0}(b,a)$ which measures the probability that a survivor in size bin a at the beginning of the previous year will grow to bin b at the beginning of the current year (columns index initial size and rows index subsequent size).[7] Growth probabilities do not include any adjustments for mortality and are applied to surviving scallops based on their original size in the preceding year.

Seasonal growth patterns are accommodated in some calculations under Option 3 (see above). Each CASA model data file contains 13 growth matrices: one matrix for annual growth (January 1 to December 31) and one matrix for growth to the middle of each month (e.g. January 1 to mid-February, January 1 to mid-March, etc.). Growth matrices are identified using the subscripts 0 to 12, where 0 is for the annual growth matrix, 1 for growth between January 1 and mid-February, 2 is for growth between January 1 and mid-March, etc. Under Option 3, in fitting to survey size composition data as an example, the program decides which growth matrix to use based on the Julian date of the survey. The monthly growth matrices are ignored under growth Options 1. All input growth matrices are ignored under Option 2 when the annual growth matrix is calculated internally based on raw shell increment data. Under Option 2:

$$P_0(b,a) = \frac{n(b \mid a)}{\sum_{j=a}^{n_L} n(j \mid a)}$$

where $n(b|a)$ is the number of individuals that started at size a and grew to size b after one year in the raw size increment data.

Age is not considered in model calculations, although age may be inferred during output calculations assuming an underlying von Bertalanffy growth curve. Two von Bertalanffy growth parameters (and K) are included in model input. The growth parameter L_∞ is not estimable in the current model because it is used in defining length bins prior to the parameter estimation phase.[8] The von Bertalanffy growth parameter K is implemented as an estimable parameter but should not be estimated because it has no effect on the objective function in the model.

The input file contains information equivalent to the von Bertalanffy growth parameter t_0 (hypothetical size at age zero) but this information does not affect the objective function in the model. Instead of entering t_0, the user enters the size at some specified age. In other words, the

[7] For clarity in bookkeeping, mortality and annual growth calculations are always based on the size on January 1.
[8] "Estimable" means a potentially estimable parameter that is specified as a variable that may be estimated in the CASA computer program. In practice, estimability depends on the available data and other factors. It may be necessary to fix certain parameters at assumed fix values or to use constraints of prior distributions for parameters that are difficult to estimate, particularly if data are limited.

user should input any age $a \geq 0$ and the corresponding a at age a on January 1. The conventional von Bertalanffy t_0 parameter is then calculated:

$$t_0 = \ln\left(1 - L / L_\infty\right) / K + a$$

Note that the calculated the calculated t_0=t_0 if a=0 and L=t_0.

Abundance, recruitment and mortality

Population abundance in each length bin during the first year of the model is:

$$N_{1,L} = N_1 \pi_{1,L}$$

where L is the size bin, and $\pi_{1,L}$ is the initial population length composition expressed as proportions so that $\sum_{L=1}^{n_L} \pi_L = 1$. $N_1 = e^{\eta}$ is total abundance at the beginning of the first modeled year and η is an estimable parameter. It is not necessary to estimate recruitment in the first year because recruitment is implicit in the product of N_1 and π_L. The current implementation of CASA takes the initial population length composition as data supplied by the user, typically based on survey size composition data and a preliminary estimate of survey size-selectivity.

Abundance at length in years after the first is calculated:

$$N_{y+1} = P_0\left(N_y \quad S_y\right) + R_{y+1}$$

where N_y is a vector (length n_L) of abundance in each length bin during year y, $\boldsymbol{P_0}$ is the matrix (n_L x n_L) of annual growth probabilities $\boldsymbol{P_0(b,a)}$, S_y is a vector of length- specific survival fractions for year y, is the operator for an element-wise product , and R_y is a vector holding length-specific abundance of new recruits at the beginning of year y.

Survival fractions are:

$$S_{y,L} = e^{-Z_{y,L}} = e^{-\left(M_{y,L} + F_{y,L} + I_{y,L}\right)}$$

where $Z_{y,L}$ is the total instantaneous mortality rate and $M_{y,L}$ is the instantaneous rate for natural mortality (see below). Length-specific fishing mortality rates are $F_{y,L}= F_y\, s_{y,L}$ where $s_{y,L}$ is the size-specific selectivity[9] for fishing in year y (scaled to a maximum of one at fully recruited size groups), F_y is the fishing mortality rate on fully selected individuals. Fully recruited fishing mortality rates are $F_y = e^{\phi + \delta_y}$ where ϕ is an estimable parameter for the log of the geometric mean of fishing mortality in all years, and δ_y is an estimable "dev" parameter.[10] The instantaneous rate for "incidental" mortality ($I_{y,L}$) accounts for mortality due to contact with the fishing gear that does not result in any catch on deck (see below).[11] The degree of variability in dev parameters for fishing mortality, natural mortality and for other variables can be controlled by specifying variances or likelihood weights • 1, as described below.

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[9] In this context, "selectivity" describes the combined effects of all factors that affect length composition of catch or landings. These factors include gear selectivity, spatial overlap of the fishery and population, size-specific targeting, size-specific discard, etc.

[10] Dev parameters are a special data type for estimable parameters in AD-Model Builder. Each set of dev parameters (e.g. for all recruitments in the model) is constrained to sum to zero. Because of the constraint, the sums $\phi + \delta_y$ involving n_y+1 terms amount to only n_y parameters.

[11] . See the section on per recruit modeling below for formulas used to relate catch, landings and indicental mortality.

Natural mortality rates $M_{y,L} = u_L e^{\zeta+\xi_y}$ may vary from year to year and by length. Variability among length groups is based on a user-specified vector u that describes the relative natural mortality rate for each length group in the model. The user supplies a value for each length group which the model rescales so that the average of all of the values is one (i.e. u is set by the user and cannot be estimated). Temporal variability in natural morality rates are modeled in the same manner as temporal variability in fishing mortality. In particular, ζ is an estimable parameter measuring the mean log natural mortality rate during all years and ξ_y is an estimable year-specific dev parameter. Several approaches are available for estimating natural mortality parameters (i.e. natural mortality covariates and surveys that measure numbers of dead individuals, see below).

Incidental mortality $I_{y,L} = F_y u_L i$ is the product of fully recruited fishing mortality (F_y, a proxy for effective fishing effort, although nominal fishing effort might be a better predictor of incidental mortality), relative incidental mortality at length (u_L) and a scaling parameter i, both of which are supplied by the user and not estimable in the model. Incidental mortality at length is supplied by the user as a vector (u) containing a value for each length group in the model. The model rescales the relative mortality vector so that the mean of the series is one.

Given abundance in each length group, natural mortality, and fishing mortality, predicted fishery catch-at-length in numbers is:

$$C_{y,L} = \frac{F_{y,L}\left(1-e^{-Z_{y,L}}\right)N_{L,y}}{Z_{y,L}}$$

Total catch number during each year is $C_y = \sum_{j=1}^{n_L} C_{y,L}$. Catch data (in weight, numbers or as length composition data) are understood to include landings (L_y) and discards (d_y) but to exclude losses to incidental mortality (i.e. $C_y = L_y + d_y$).

Discard data are supplied by the user in the form of discarded biomass in each year or a discard rate for each year (or a combination of biomass levels and rates). In the current model, discards have the same selectivity as landed catch and size composition data for discards are not included in the input file.[12] It is important to remember that discard rates in CASA are defined the ratio of discards to landings (d/L). The user may also specify a mortal discard fraction between zero and one if some discards survive. If the discard fraction is less than one, then the discarded biomass and discard rates in the model are reduced correspondingly. See the section on per recruit modeling below for formulas used to relate catch, landings and incidental mortality.

Recruitment (the sum of new recruits in all length bins) at the beginning of each year after the first is calculated:

$$Ry = e^{\rho+\gamma_y}$$

where ρ is an estimable parameter that measures the geometric mean recruitment and the γ_y are estimable dev parameters that measure inter-annual variability in recruitment. As with natural mortality devs, the user specified variance or likelihood weight • 1 can be used to help estimate recruitment deviations (see below).

[12] The model will be modified in future to model discards and landing separately, and to use size composition data for discards.

Proportions of recruits in each length group are calculated based on a beta distribution *B(w,r)* over the first n_r length bins that is constrained to be concave down.[13] Proportions of new recruits in each size group are the same from year to year. Beta distribution coefficients must be larger than one for the shape of the distribution to be unimodal. Therefore, $w=1+e^{\omega}$ and $r=1+e^{\rho}$, where ω and ρ are estimable parameters. It is presumably better to calculate the parameters in this manner than as bounded parameters because there is likely to be less distortion of the Hessian for *w* and *r* values close to one and parameter estimation is likely to be more efficient.

Surplus production during each year of the model can be computed approximately from biomass and catch estimates (Jacobson et al., 2002):

$$P_t = B_{t+1} - B_t + C_t$$

In future versions of the CASA model, surplus production will be more calculated more accurately by projecting the population at the beginning of the year forward one year assuming only natural mortality.

Weight at length[14]

The assumed body weight for size bins except the last is calculated using user-specified length-weight parameters and the middle of the size group. Different length-weight parameters are used for the population and for the commercial fishery. Mean body weight in the last size bin is read from the input file and can vary from year to year. Typically, mean weight in the last size bin for the population would be computed based on survey length composition data for large individuals and the population length –weight relationship. Mean weight in the last size bin for the fishery would be computed in the same manner based on fishery size composition data.

In principle, these calculations could be carried out in the model itself because all of the required information is available. In practice, it seems better to do the calculations externally and supply them to the model as inputs because of decisions that typically have to be made about smoothing the estimates and years with missing data.

Population summary variables

Total abundance at the beginning of the year is the sum of abundance at length $N_{y,L}$ at the beginning of the year. Average annual abundance for a particular length group is:

$$\overline{N}_{y,L} = N_{y,L} \frac{1-e^{-Z_{y,L}}}{Z_{y,L}}$$

The current implementation of the assessment model assumes different weight-at-length relationships for the stock and the fishery. Average stock biomass is computed using the population weight at length information.

Total stock biomass is:

$$B_y = \sum_{L=1}^{n_L} N_{y,L} w_L$$

[13] Standard beta distributions used to describe recruit size distributions and in priors are often constrained to be unimodal in the CASA model. Beta distributions *B(w,r)* with mean $\mu = w/w+r$ and variance $\sigma^2 = wr/\left[(w+r)^2(w+r+1)\right]$ are unimodal when w > 1 and r >1. See http://en.wikipedia.org/wiki/Beta_distribution for more information.

[14] Model input data include a scalar that is used to convert the units for length-weight parameters (e.g. grams) to the units of the biomass estimates and landings data (e.g. mt).

where w_L is weight at length for the population on January 1. Total catch weight is:

$$W_y = \sum_{L=1}^{n_L} C_{y,L} w'_L$$

where w'_L is weight at length in the fishery.

F_y estimates for two years are comparable only when the fishery selectivity in the model was the same in both years. A simpler exploitation index is calculated for use when fishery selectivity changes over time:

$$U_y = \frac{C_y}{\sum_{j=x}^{n_L} N_{y,L}}$$

where x is a user-specified length bin (usually at or below the first bin that is fully selected during all fishery selectivity periods). U_y exploitation indices from years with different selectivity patterns may be relatively comparable if x is chosen carefully.

Spawner abundance in each year is (T_y) is computed:

$$T_y = \sum_{L=1}^{n_L} N_{y,L} e^{-\tau Z_y} g_L$$

Where $0 \le \tau \le 1$ is the fraction of the year elapsed before spawning occurs (supplied by the user). Maturity at length (g_L) is from an ascending logistic curve:

$$g_L = \frac{1}{1+e^{a-bL}}$$

with parameters a and b supplied by the user. Spawner biomass is computed using the population length-weight vaoues.

Egg production (S_y) in each year is computed:

$$S_y = \sum_{L=1}^{n_L} N_{y,L} e^{-\tau Z_y} g_L x_L$$

where:

$$x_L = cL^v$$

Where the fecundity parameters (c and v) for fecundity are supplied by the user. Fecundity parameters per se include no adjustments for maturity or survival. They should represent reproductive output for a spawner of given size.

Fishery and survey selectivity

The current implementation of CASA includes six options for calculating fishery and survey selectivity patterns. Fishery selectivity may differ among “fishery periods” defined by the user. Selectivity patterns that depend on length are calculated using lengths at the mid-point of each bin ( ). After initial calculations (described below), selectivity curves are rescaled to a maximum value of one.

Option 1 is a flat with s_L=1 for all length bins. Option 2 is an ascending logistic curve:

$$s_{y,} = \frac{1}{1+e^{A_Y - B_Y}}$$

Option 3 is an ascending logistic curve with a minimum asymptotic minimum size for small size bins on the left.

$$s_{y,} = \left(\frac{1}{1+e^{A_Y - B_Y}}\right)(1-D_y)+D_y$$

Option 4 is a descending logistic curve:

$$s_{y,} = 1 - \frac{1}{1+e^{A_Y - B_Y}}$$

Option 5 is a descending logistic curve with a minimum asymptotic minimum size for large size bins on the right:

$$s_{y,} = \left(1 - \frac{1}{1+e^{A_Y - B_Y}}\right)(1-D_y)+D_y$$

Option 6 is a double logistic curve used to represent "domed-shape" selectivity patterns with highest selectivity on intermediate size groups:

$$s_{y,} = \left(\frac{1}{1+e^{A_Y - B_Y}}\right)\left(1 - \frac{1}{1+e^{D_Y - G_Y}}\right)$$

The coefficients for selectivity curves A_Y, B_Y, D_Y and G_Y carry subscripts for time because they may vary between fishery selectivity periods defined by the user. All options are parameterized so that the coefficients A_Y, B_Y, D_Y and G_Y are positive. Under options 3 and 5, D_y is a proportion that must lie between 0 and 1.

Depending on the option, estimable selectivity parameters may include α, β, δ and γ. For options 2, 4 and 6, $A_Y = e^{\alpha_Y}$, $B_Y = e^{\beta_Y}$, $D_Y = e^{\delta_Y}$ and $G_Y = e^{\gamma_Y}$. Options 3 and 5 use the same conventions for A_Y and B_Y, however, the coefficient D_Y is a proportion estimated as a logit-transformed parameter (i.e. $\delta_Y = \ln[D_Y/(1-D_y)]$) so that:

$$D_Y = \frac{e^{\delta_Y}}{1+e^{\delta_Y}}$$

The user can choose, independently of all other parameters, to either estimate each fishery selectivity parameter or to keep it at its initial value. Under Option 2, for example, the user can estimate the intercept α_Y, while keep the slope β_Y at its initial value.

Per recruit recruit modeling

The per recruit model in CASA uses the same population model as in other model calculations under conditions identical to the last year in the model. It is a standard length-based approach except that discard and incidental mortality are accommodated in all calculations. In per recruit calculations, fishing mortality rates and associated yield estimates are understood to include landings and discard mortality, but to exclude incidental mortality. Thus, landings per recruit L are:

$$L = \frac{C}{(1+\Delta)}$$

where C is total catch (yield) per recruit and Δ is the ratio of discards D to landings in the last year of the model. Discards per recruit are calculated:

$$D = \Delta L$$

Losses due to incidental mortality (G) are calculated:

$$G = \frac{I\left(1 - e^{-Z}\right)B}{Z}$$
$$= IK$$

where $I = F\,u$ is the incidental mortality rate, u is a user-specified multiplier (see above) and B is stock biomass per recruit. Note that $C=FK$ so that $K=C/F$. Then,

$$G = \frac{FuC}{F}$$
$$G = uC$$

The model will estimate a wide variety ($F_{\%SBR}$, F_{max} and $F_{0.1}$) of per recruit model reference points as parameters. For example,

$$F_{\%SBR} = e^{\theta_j}$$

where $F_{\%SBR}$ is the fishing mortality reference point that provides a user specified percentage of maximum SBR. θ_j is the model parameter for the j^{th} reference point.

A complete per recruit output table is generated in all model runs that can be used for evaluating the shape of YPR and SBR curves, including the existence of particular reference points.

Per recruit reference points are time consuming to estimate and it is usually better to estimate them after other more important population dynamics parameters are estimated. Phase of estimation can be controlled individually for %SBR, F_{MAX} and $F_{0.1}$ so that per recruit calculations can be delayed as long as possible. If the phase is set to zero or a negative integer, then the reference point will not be estimated. As described below, estimation of F_{max} always entails an additional phase of estimation. For example, if the phase specified for F_{max} is 2, then the parameter will be estimated initially in phase 2 and finalized the last phase (phase >= 3). This is done so that the estimate from phase 2 can be used as an initial value in a slightly different goodness of fit calculation during the latter phase.

Per recruit reference points should have no effect on other model estimates. Residuals (calculated – target) for %SBR, $F_{0.1}$ and F_{max} reference points should always be very close to zero. Problems may arise, however, if reference points (particularly F_{max}) fall on the upper bound for fishing mortality. In such cases, the model will warn the user and advise that the offending reference points should not be estimated. *It is good practice to run CASA with reference point calculations turned on and then off to see if biomass and fishing mortality estimates change*.

The user specifies the number of estimates required and the target %SBR level for each. For example, the target levels for four %SBR reference points might be 0.2, 0.3, 0.4 and 0.5 to estimate $F_{20\%}$, $F_{30\%}$, $F_{40\%}$ and $F_{50\%}$. The user has the option of estimating F_{max} and/or $F_{0.1}$ as model parameters also but it is not necessary to supply target values.

Tuning and goodness of fit

There are two steps in calculating the negative log likelihood (NLL) used to measure how well the model fits each type of data. The first step is to calculate the predicted values for data. The second step is to calculate the NLL of the data given the predicted value. The overall goodness of fit measure for the model is the weighted sum of NLL values for each type of data and each constraint:

$$\Lambda = \sum \lambda_j L_j$$

where λ_j is a weighting factor for data set j (usually λ_j=1, see below), and L_j is the NLL for the data set. The NLL for a particular data is itself is usually a weighted sum:

$$L_j = \sum_{i=1}^{n_j} \psi_{j,i} L_{j,i}$$

where n_j is the number of observations, $\psi_{j,i}$ is an observation-specific weight (usually $\psi_{j,i}$ =1, see below), and $L_{j,i}$ is the NLL for a single observation.

Maximum likelihood approaches reduce the need to specify *ad-h*oc weighting factors (λ and ϕ) for data sets or single observations, because weights can often be taken from the data (e.g. using CVs routinely calculated for bottom trawl survey abundance indices) or estimated internally along with other parameters. In addition, robust maximum likelihood approaches (see below) may be preferable to simply down-weighting an observation or data set. However, despite subjectivity and theoretical arguments against use of *ad-hoc* weights, it is often useful in practical work to manipulate weighting factors, if only for sensitivity analysis or to turn an observation off entirely. Observation specific weighting factors are available for most types of data in the CASA model.

Missing data

Availability of data is an important consideration in deciding how to structure a stock assessment model. The possibility of obtaining reliable estimates will depend on the availability of sufficient data. However, NLL calculations and the general structure of the CASA model are such that missing data can usually be accommodated automatically. With the exception of catch data (which must be supplied for each year, even if catch was zero), the model calculates that NLL for each datum that is available. No NLL calculations are made for data that are not available and missing data do not generally hinder model calculations.

Likelihood kernels

Log likelihood calculations in the current implementation of the CASA model use log likelihood "kernels" or "concentrated likelihoods" that omit constants. The constants can be omitted because they do not affect slope of the NLL surface, final point estimates for parameters or asymptotic variance estimates.

For data with normally distributed measurement errors, the complete NLL for one observation is:

$$L = \ln(\sigma) + \ln\left(\sqrt{2\pi}\right) + 0.5\left(\frac{x-u}{\sigma}\right)^2$$

The constant $\ln\left(\sqrt{2\pi}\right)$ can always be omitted. If the standard deviation is known or assumed known, then ln(σ) can be omitted as well because it is a constant that does not affect derivatives. In such cases, the concentrated NLL is:

$$L = 0.5\left(\frac{x-\mu}{\sigma}\right)^2$$

If there are N observations with possible different variances (known or assumed known) and possibly different expected values:

$$L = 0.5\sum_{i=1}^{N}\left(\frac{x_i-\mu_i}{\sigma_i}\right)^2$$

If the standard deviation for a normally distributed quantity is not known and is estimated (implicitly or explicitly) by the model, then one of two equivalent calculations is used. Both approaches assume that all observations have the same variance and standard deviation. The first approach is used when all observations have the same weight in the NLL:

$$L = 0.5N\ln\left[\sum_{i=1}^{N}(x_i - u)^2\right]$$

The second approach is equivalent but used when the weights for each observation (w_i) may differ:

$$L = \sum_{i=1}^{N} w_i\left[\ln(\sigma) + 0.5\left(\frac{x_i - u}{\sigma}\right)^2\right]$$

In the latter case, the maximum likelihood estimator:

$$\hat{\sigma} = \sqrt{\frac{\sum_{i=1}^{N}(x_i - \hat{x})^2}{N}}$$

(where $\hat{x}$ is the average or predicted value from the model) is used explicitly for σ. The maximum likelihood estimator is biased by $N/(N-d_f)$ where d_f is degrees of freedom for the model. The bias may be significant for small sample sizes, which are common in stock assessment modeling, but d_f is usually unknown.

If data x have lognormal measurement errors, then ln(x) is normal and L is calculated as above. In some cases it is necessary to correct for bias in converting arithmetic scale means to log scale means (and *vice-versa*) because $\overline{x} = e^{\overline{\chi} + \sigma^2/2}$ where χ=ln(x). It is often convenient to convert arithmetic scale CVs for lognormal variables to log scale standard deviations using $\sigma = \sqrt{\ln(1 + CV^2)}$.

For data with multinomial measurement errors, the likelihood kernel is:

$$L = n\sum_{i=1}^{n} p_i \ln(\theta_i) - K$$

where n is the known or assumed number of observations (the “effective” sample size), p_i is the proportion of observations in bin i, and θ_i is the model’s estimate of the probability of an observation in the bin. For surveys, θ_i is adjusted for mortality up to the date of the survey and for growth up to the mid-point of the month in which the survey occurs. For fisheries, θ_i accommodates all of the mortality during the current year and is adjusted for growth during January 1 to mid-July. The constant K is used for convenience to make L easier to interpret. It measures the lowest value of L that could be achieved if the data fit matched the model’s expectations exactly:

$$K = n\sum_{i=1}^{n} p_i \ln(p_i)$$

For data x that have measurement errors with expected values of zero from a gamma distribution:

$$L = (\gamma - 1)\ln\left(x/\beta\right) - x/\beta - \ln(\beta)$$

where β>0 and γ>0 are gamma distribution parameters in the model. For data that lie between zero and one with measurement errors from a beta distribution:

$$L = (p-1)\ln(x) + (q-1)\ln(1-x)$$

where p>0 and q>0 are parameters in the model.

In CASA model calculations, distributions are usually described in terms of the mean and CV. Normal, gamma and beta distribution parameters can be calculated mean and CV by the method of moments.[15] Means, CV's and distributional parameters may, depending on the situation, be estimated in the model or specified by the user.

The NLL for a datum x from gamma distribution is:

$$L = (1-k)*\ln(x) + \frac{x}{\theta} + \ln[\Gamma(k)] + k\ln(\theta)$$

where k is the shape parameter and θ is the scale parameter. The last two terms on the right are constants and can be omitted if k and θ are not estimated. Under these circumstances,

$$L = (1-k)*\ln(x) + \frac{x}{\theta}$$

Robust methods

Goodness of fit for survey data may be calculated using a "robust" maximum likelihood method instead of the standard method that assumes lognormal measurement errors. The robust method may be useful when survey data are noisy or include outliers.

Robust likelihood calculations in CASA assume that measurement errors are from a Student's t distribution with user-specified degrees of freedom d_f. Degrees of freedom are specified independently for each observation so that robust calculations can be carried out for as many (or as few) cases as required. The t distribution is similar to the normal distribution for $d_f \geq 30$. As d_f is reduced, the tails of the t distribution become fatter so that outliers have higher probability and less effect on model estimates. If $d_f = 0$, then measurement errors are assumed in the model to be normally distributed.

The first step in robust NLL calculations is to standardize the measurement error residual $t = (x - \bar{x})/\sigma$ based on the mean and standard deviation. Then:

$$L = \ln\left(1 + \frac{t^2}{d_f}\right)\left(1 - \frac{1-d_f}{2}\right) - \frac{\ln(d_f)}{2}$$

Catch weight data

Catch data (landings plus discards) are assumed to have normally distributed measurement errors with a user specified CV. The standard deviation for catch weight in a particular year is $\sigma_Y = \kappa\hat{C}_y$ where "^" indicates that the variable is a model estimate and errors in catch are assumed to be normally distributed. The standardized residual used in computing NLL for a single catch observation and in making residual plots is $r_Y = \left(C_Y - \hat{C}_Y\right)/\sigma_Y$.

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[15] Parameters for standard beta distributions $B(w,r)$ with mean $\mu = w/w + r$ and variance $\sigma^2 = wr/\left[(w+r)^2(w+r+1)\right]$ are calculated from user-specified means and variances by the method of moments. In particular, $w = \mu\left[\mu(1-\mu)/\sigma^2 - 1\right]$ and $r = (1-\mu)\left[\mu(1-\mu)/\sigma^2 - 1\right]$. Not all combinations of μ and σ^2 are feasible. In general, a beta distribution exists for combinations of μ and σ^2 if $0 < \mu < 1$ and $0 < \sigma^2 < \mu(1-\mu)$. Thus, for a user-specified mean μ between zero and one, the largest feasible variance is $\sigma^2 < \mu(1-\mu)$. These conditions are used in the model to check user-specified values for μ and σ^2. See http://en.wikipedia.org/wiki/Beta_distribution for more information.

Specification of landings, discards, catch

Landings, discard and catch data are in units of weight and are for a single or "composite" fishery in the current version of the CASA model. The estimated fishery selectivity is assumed to apply to the discards so that, in effect, the length composition of catch, landings and discards are the same.

Discards are from external estimates (d_t) supplied by the user. If $d_t \geq 0$, then the data are used as the ratio of discard to landed catch so that:

$$D_t = L_t \Delta_t$$

where $\Delta_t = D_t/L_t$ is the ratio of discard and landings (a.k.a. d/K ratios) for each year. If $d_t < 0$ then the data are treated as discard in units of weight:

$$D_t = abs(d_t).$$

In either case, total catch is the sum of discards and landed catch ($C_t = L_t + D_t$). It is possible to use discards in weight $d_t < 0$ for some years and discard as proportions $d_t > 0$ for other years in the same model run.

If catches are estimated (see below) so that the estimated catch $\hat{C}_t$ does not necessarily equal observed landings plus discard, then estimated landings are computed:

$$\hat{L}_t = \frac{\hat{C}_t}{1 + \Delta_t}$$

Estimated discards are:

$$\hat{D}_t = \Delta_t \hat{L}_t.$$

Note that $\hat{C}_t = \hat{L}_t + \hat{D}_t$ as would be expected.

Fishery length composition data

Data describing numbers or relative numbers of individuals at length in catch data (fishery catch-at-length) are modeled as multinomial proportions $c_{y,L}$:

$$c_{y,L} = \frac{C_{y,L}}{\sum_{j=1}^{n_L} C_{y,j}}$$

The NLL for the observed proportions in each year is computed based on the kernel for the multinomial distribution, the model's estimate of proportional catch-at-length $(\hat{c}_Y)$ and an estimate of effective sample size ${}^{C}N_Y$ supplied by the user. Care is required in specifying effective sample sizes, because catch-at-length data typically carry substantially less information than would be expected based on the number of individuals measured. Typical conventions make ${}^{c}N_Y \leq 200$ (Fournier and Archibald, 1982) or set ${}^{C}N_Y$ equal to the number of trips or tows sampled (Pennington et al., 2002). Effective sample sizes are sometimes chosen based on goodness of fits in preliminary model runs (Methot, 2000; Butler et al., 2003).

Standardized residuals are not used in computing NLL fishery length composition data. However, approximate standardized residuals $r_y = (c_{y,L} - \hat{c}_{y,L})/\sigma_{y,L}$ with standard deviations $\sigma_{y,L} = \sqrt{\hat{c}_{y,L}(1 - \hat{c}_{y,L})/{}^{c}N_y}$ based on the theoretical variance for proportions are computed for use in making residual plots.

Survey index data

In CASA model calculations, “survey indices” are data from any source that reflect relative proportional changes in an underlying population state variable. In the current version, surveys may measure stock abundance at a particular point in time (e.g. when a survey was carried out), stock biomass at a particular point in time, or numbers of animals that dies of natural mortality during a user-specified period. For example, the first option is useful for bottom trawl surveys that record numbers of individuals, the second option is useful for bottom trawl surveys that record total weight, and the third option is useful for survey data that track trends in numbers of animals that died due to natural mortality (e.g. survey data for sea scallop “clappers”). Survey data that measure trends in numbers dead due to natural mortality can be useful in modeling time trends in natural mortality. In principle, the model will estimate model natural mortality and other parameters so that predicted numbers dead and the index data match in either relative or absolute terms.

In the current implementation of the CASA model, survey indices are assumed to be linear indices of abundance or biomass so that changes in the index (apart from measurement error) are assumed due to proportional changes in the population. Nonlinear commercial catch rate data are handled separately (see below). Survey index and fishery length composition data are handled separately from trend data (see below). Survey data may or may not have corresponding length composition information.

In general, survey index data give one number that summarizes some aspect of the population over a wide range of length bins. Selectivity parameters measure the relative contribution of each length bin to the index. Options and procedures for estimating survey selectivity patterns are the same as for fishery selectivity patterns, but survey selectivity patterns are not allowed to change over time.

NLL calculations for survey indices use predicted values calculated:

$$\hat{I}_{k,y} = q_k A_{k,y}$$

where q_k is a scaling factor for survey index k, and $A_{k,y}$ is stock available to the survey. The scaling factor is computed using the maximum likelihood estimator:

$$q_k = e^{\frac{\sum_{i=1}^{N_k}\left[\ln\left(\frac{I_{k,i}}{A_{k,i}}\right) \big/ \sigma^2_{k,i}\right]}{\sum_{j=1}^{N_k}\left(1 \big/ \sigma^2_{k,j}\right)}}$$

where N_v and is the log scale variance corresponding to the assumed CV for the survey observation.[16]

Available stock for surveys measuring trends in abundance or biomass is calculated:

$$A_{k,y} = \sum_{L=1}^{n_L} s_{k,L} N_{y,L} e^{-Z_{y,L}\tau_{k,y}}$$

[16] Scaling factors in previous versions were calculated $q_s = e^{\varpi_s}$ where ϖ_s is an estimable and survey-specific parameter. However, prior distributions were shown to have a strong effect on the parameters such that the relationship $N=qA$ did not hold. The approach in the current model avoids this problem.

where $s_{k,L}$ is size-specific selectivity of the survey, $\tau_{k,y}=J_{k,y}/365$, $J_{k,y}$ is the Julian date of the survey in year y, and $e^{-Z_y \tau_{k,y}}$ is a correction for mortality prior to the survey. Available biomass is calculated in the same way except that body weights w_L are included in the product on the right hand side.

Available stock for indices that track numbers dead by natural mortality is:

$$A_{k,y} = \sum_{L=1}^{n_L} s_{k,L} \tilde{M}_{y,L} \overline{N}_{y,L}$$

where $\overline{N}_{y,L}$ is average abundance during the user-specified period of availability and $\tilde{M}_{y,L}$ is the instantaneous rate of natural mortality for the period of availability. Average abundance during the period of availability is:

$$\overline{N}_{y,L} = \frac{\tilde{N}_{y,L}\left(1-e^{-\tilde{Z}_{y,L}}\right)}{\tilde{Z}_{y,L}}$$

where $\tilde{N}_{y,L} = N_{y,L} e^{-Z\Delta}$ is abundance at elapsed time of year $\Delta=\tau_{k,y}-\nu_k$, $\nu_k=j_k/365$, and j_k is the user-specified duration in days for the period of availability. The instantaneous rates for total $\tilde{Z}_{y,L} = Z_{y,L}\left(\tau_{k,y}-\nu_k\right)$ and natural $\tilde{M}_{y,L} = M_{y,L}\left(\tau_{k,y}-\nu_k\right)$ mortality are also adjusted to correspond to the period of availability. In using this approach, the user should be aware that the length based selectivity estimated by the model for the dead animal survey ($s_{k,L}$) is conditional on the assumed pattern of length-specific natural mortality (u) which was specified as data in the input file.

NLL calculations for survey index data assume that log scale measurement errors are either normally distributed (default approach) or from a *t* distribution (robust estimation approach). In either case, log scale measurement errors are assumed to have mean zero and log scale standard errors either estimated internally by the model or calculated from the arithmetic CVs supplied with the survey data.

The standardized residual used in computing NLL for one survey index observation is $r_{k,y} = \ln\left(I_{k,y} / \hat{I}_{k,y}\right) / \sigma_{k,y}$ where $I_{k,y}$ is the observation. The standard deviations $\sigma_{k,y}$ will vary among surveys and years if CVs are used to specify the variance of measurement errors. Otherwise a single standard deviation is estimated internally for the survey as a whole.

Survey length composition data

Length bins for fishery and survey length composition data are flexible and the flexibility affects goodness of fit calculations in ways that may be important to consider in some applications. The user specifies the starting size (bottom of first bin) and number of bins used for each type of fishery and survey length composition. The input data for each length composition record identifies the first/last length bins to be used and whether they are plus groups that should include all smaller/larger length groups in the data and population model when calculating goodness of fit. Goodness of fit calculations are carried out over the range of lengths specified by the user. Thus length data in the input file may contain large or small size bins that are ignored in goodness of fit calculations. As described above, the starting size and bin size for the population model are specified separately. In the ideal and simplest case, the minimum size and same length bins are used for the population and for all length data. However, as described below, length specifications in data and the population model may differ.

For example, the implicit definitions of plus groups in the model and data may differ. If the first bin used for length data is a plus group, then the first bin will contain the sum of length data from the corresponding and smaller bins of the original length composition record. However, the first bin in the population model is never a plus group. Thus, predicted values for a plus group will contain the sum of the corresponding and smaller bins in the population. The observed and predicted values will not be perfectly comparable if the starting sizes for the data and population model differ. Similarly, if the last bin in the length data is a plus group, it will contain original length composition data for the corresponding and all larger bins. Predicted values for a plus group in the population will be the sum for the corresponding bin and all larger size groups in the population, implicitly including sizes > L_{∞}.. The two definitions of the plus group will differ and goodness of fit calculation may be impaired if the original length composition data does not include all of the large individuals in samples.

In the current version of the CASA model, the size of length composition bins must be • L_{bin} in the population model (this constraint will be removed in later versions). Ideally, the size of data length bins is the same or a multiple of the size of length bins in the population. However, this is not required and the model will prorate the predicted population composition for each bin into adjacent data bins when calculating goodness of fit. With a 30-34 mm population bin and 22-31and 32-41 mm population bins, for example, the predicted proportion in the population bin would be prorated so that 2/5 was assigned to the first data bin and 3/5 was assigned to the second data bin. This proration approach is problematic when it is used to prorate the plus group in the population model into two data bins because it assumes that abundance is uniform over lengths within the population group. The distribution of lengths in a real population might be far from uniform between the assumed upper and lower bounds of the plus group.

The first bin in each length composition data record must be • L_{min} which is the smallest size group in the population model. If the last data bin is a plus group, then the *lower* bound of the last data bin must be • the upper bound of the last population bin. Otherwise, if the last data bin is not a plus group, the *upper* bound of the last data bin must be • the upper bound of the population bin.

NLL calculations for survey length composition data are similar to calculations for fishery length composition data. Surveys index data may measure trends in stock abundance or biomass but survey length composition data are always for numbers (not weight) of individuals in each length group. Survey length composition data represent a sample from the true stock which is modified by survey selectivity, sampling errors and, if applicable, errors in recording length data. For example, with errors in length measurements, individuals belonging to length bin j, are mistakenly assigned to adjacent length bins j-2, j-1, j+1 or j+2 with some specified probability. Well-tested methods for dealing with errors in length data can be applied if some information about the distribution of the errors is available (e.g. Methot 2000).

Prior to any other calculations, observed survey length composition data are converted to multinomial proportions:

$$i_{k,y,L} = \frac{n_{k,y,L}}{\sum_{j=L_{k,y}^{first}}^{L_{k,y}^{last}} n_{k,y,j}}$$

where $n_{k,y,j}$ is an original datum and $i_{k,y,L}$ is the corresponding proportion. As described above, the user specifies the first $L_{k,y}^{first}$ and last $L_{k,y}^{last}$ length groups to be used in calculating goodness of fit for each length composition and specifies whether the largest and smallest groups should be treated as "plus" groups that contain all smaller or larger individuals.

Using notation for goodness of fit survey index data (see above), predicted length compositions for surveys that track abundance or biomass are calculated:

$$A_{k,y,L} = \frac{s_{k,L} N_{y,L} e^{-Z_{y,j}\tau_{k,y}}}{\sum_{L=L_{k,y}^{first}}^{L_{k,y}^{last}} s_{k,j} N_{y,j} e^{-Z_{y,j}\tau_{k,y}}}$$

Predicted length compositions for surveys that track numbers of individuals killed by natural mortality are calculated:

$$A_{k,y} = \frac{s_{k,L} \tilde{M}_{y,L} \overline{N}_{y,L}}{\sum_{L=L_{k,y}^{first}}^{L_{k,y}^{Last}} s_{k,L} \tilde{M}_{y,L} \overline{N}_{y,L}}$$

Considering the possibility of structured measurement errors, the expected length composition $A'_{k,y}$ for survey catches is:

$$A'_{k,y} = A_{k,y} E_k$$

where E_k is an error matrix that simulates errors in collecting length data by mapping true length bins in the model to observed length bins in the data.

The error matrix E_k has n_L rows (one for each true length bin) and n_L columns (one for each possible observed length bin). For example, row k and column j of the error matrix gives the conditional probability P($k|j$) of being assigned to bin k, given that an individual actually belongs to bin j. More generally, column j gives the probabilities that an individual actually belonging to length bin j will be recorded as being in length bins j-2, j-1, j, j+1, j+2 and so on. The columns of E_k add to one to account for all possible outcomes in assigning individuals to observed length bins. E_k is the identity matrix if there are no structured measurement errors. In CASA, the probabilities in the error matrix are computed from a normal distribution with mean zero and $CV = e^{\pi_k}$, where π_k is an estimable parameter. The normal distribution is truncated to cover a user-specified number of observed bins (e.g. 3 bins on either side of the true length bin).

The NLL for observed proportions at length in each survey and year is computed with the kernel for a multinomial distribution, the model's estimate of proportional survey catch-at-length $\left(\hat{i}_{k,y,L}\right)$ and THE effective sample size ${}^{I}N_{Y}$ supplied by the user. Standardized residuals for residual plots are computed as for fishery length composition data.

Effective sample size for length composition data
Effective sample sizes that are specified by the user are used in goodness of fit calculations for survey and fishery length composition data. A post-hoc estimate of effective sample size can be calculated based on goodness of fit in a model run (Methot 1989). Consider the variance of residuals for a single set of length composition data with N bins used in calculations. The variance of the sum based on the multinomial distribution is:

$$\sigma^2 = \sum_{j=1}^{N}\left[\frac{\hat{p}_j\left(1-\hat{p}_j\right)}{\varphi}\right]$$

where φ is the effective sample size for the multinomial and $\overline{p}_j$ is the predicted proportion in the j^{th} bin from the model run. Solve for φ to get:

$$\varphi = \frac{\sum_{j=1}^{N}\left[\hat{p}_j\left(1-\hat{p}_j\right)\right]}{\sigma^2}$$

The variance of the sum of residuals can also be calculated:

$$\sigma^2 = \sum_{j=1}^{N}\left(p_j - \hat{p}_j\right)^2$$

This formula is approximate because it ignores the traditional correction for bias. Substitute the third expression into the second to get:

$$\varphi = \frac{\sum_{j=1}^{N}\left[\hat{p}_j\left(1-\hat{p}_j\right)\right]}{\sum_{k=1}^{N}\left(p_j - \hat{p}_j\right)^2}$$

which can be calculated based on model outputs. The assumed and effective sample sizes will be similar in a reasonable model when the assumed sample sizes are approximately correct. Effective sample size calculations can be used iteratively to manually adjust input vales to reasonable levels (Methot 1989).

Variance constraints on dev parameters
Variability in dev parameters (e.g. for natural mortality, recruitment or fishing mortality) can be limited using variance constraints that assume the deviations are either independent or that they are autocorrelated and follow a random walk. When a variance constraint for independent deviations is activated, the model calculates the NLL for each log scale residual γ_y / σ_γ, where γ_y is a dev parameter and σ is a log-scale standard deviation. If the user supplies a positive value for the arithmetic scale CV, then the NLL is calculated assuming the variance is known. Otherwise, the user-supplied CV is ignored and the NLL is calculated with the standard deviation estimated internally. Calculations for autocorrelated deviations are the same except

that the residuals are $\left(\gamma_y - \gamma_{y-1}\right) / \sigma_\gamma$ and the number of residuals is one less than the number of dev parameters.

LPUE data

Commercial landings per unit of fishing effort (LPUE) data are modeled in the current implementation of the CASA model as a linear function of average biomass available to the fishery, and as a nonlinear function of average available abundance. The nonlinear relationship with abundance is meant to reflect limitations in "shucking" capacity for sea scallops.[17] Briefly, tows with large numbers of scallops require more time to sort and shuck and therefore reduce LPUE from fishing trips when abundance is high. The effect is exaggerated when the catch is composed of relatively small individuals. In other words, at any given level of stock biomass, LPUE is reduced as the number of individuals in the catch increases or, equivalently, as the mean size of individuals in the catch is reduced.

Average available abundance in LPUE calculations is:

$$^{a}\overline{N}_y = \sum_{L=1}^{n_L} s_{y,L} \overline{N}_{y,L}$$

and average available biomass is:

$$^{a}\overline{B}_y = \sum_{L=1}^{n_L} s_{y,L} w_L^f \overline{N}_{y,L}$$

where the weights at length w_L^f are for the fishery rather than the population. Predicted values for LPUE data are calculated:

$$\hat{L}_y = \frac{^{a}\overline{B}_y \eta}{\sqrt{\varphi^2 + {^{a}\overline{N}_y}^2}}$$

Measurement errors in LPUE data are assumed normally distributed with standard deviations $\sigma_y = CV_y L_y$. Standardized residuals are $r_y = \left(L_y - \hat{L}_y\right) / \sigma_y$.

Per recruit (SBR and YPR) reference points[18]

The user specifies a target %SBR value for each reference point that is estimated. Goodness of fit is calculated as the sum of squared differences between the target %SBR and %SBR calculated based on the reference point parameter. Except in pathological situations, it is always possible to estimate %SBR reference point parameters so that the target and calculated %SBR levels match exactly. Reference point parameters should have no effect on other model estimates and the residual (calculated – target %SBR) should always be very close to zero. Goodness of fit for $F_{0.1}$ estimates is calculated in a manner similar to %SBR reference points. Goodness of fit is calculated as the squared difference between the slope of the yield curve at the estimate and one-tenth of the slope at the origin. Slopes are computed numerically using central differences if possible or one-sided (right hand) differences if necessary.

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[17] D. Hart, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, MA, pers. comm.

[18] This approach is not currently estimated because of performance problems. The user can, however, estimate per recruit reference point from a detailed table written in the main output file (nc.rep). However, variances are not available in the table.

F_{max} is estimated differently in preliminary and final phases. In preliminary phases, goodness of fit for F_{max} is calculated as $(1/Y)^2$, where Y is yield per recruit at the current estimate of F_{max}. In other words, yield per recruit is maximized by finding the parameter estimate that minimizes it's inverse. This preliminary approach is very robust and will find F_{max} if it exists. However, it involves a non-zero residual ($1/Y$) that interferes with calculation of variances and might affect other model estimates. In final phases, goodness of fit for F_{max} is calculated as (d^2) where d is the slope of the yield per recruit curve at F_{max}. The two approaches give the same estimates of F_{MAX} but the goodness of fit approach used in the final phases has a residual of zero (so that other model estimates are not affected) and gives more reasonable variance estimates. The latter goodness of fit calculation is not used during initial phases because the estimates of F_{MAX} tend to "drift down" the right hand side of the yield curve in the direction of decreasing slope. Thus, the goodness of fit calculation used in final phases works well only when the initial estimate of F_{MAX} is very close to the best estimate.

Per recruit reference points should have little or no effect on other model estimates. Problems may arise, however, if reference points (particularly F_{max}) fall on the upper bound for fishing mortality. In such cases, the model will warn the user and advise that the offending reference points should not be estimated. *It is good practice to run CASA with and without reference point calculations to ensure that reference points do not affect other model estimates including abundance, recruitments and fishing mortality rates*.

Growth data

Growth data in CASA consist of records giving initial length, length after one year of growth, and number of corresponding observations. Growth data may be used to help estimate growth parameters that determine the growth matrix $\mathcal{P}$. The first step is to convert the data for each starting length to proportions:

$$P(b,a) = \frac{n(b,a)}{\sum_{j=n_L-b+1}^{n_L} n(j,a)}$$

where $n(b,a)$ is the number of individuals starting at size *that* grew to size b after one year. The NLL is computed assuming that observed proportions $p(a|b)$ at each starting size are a sample from a multinomial distribution with probabilities given by the corresponding column in the models estimated growth matrix $\mathcal{P}$. The user must specify an effective sample size ${}^{P}N_j$ based, for example, on the number of observations in each bin or the number of individuals contributing data to each bin. Observations outside bin ranges specified by the user are ignored. Standardized residuals for plotting are computed based on the variance for proportions.

Survey gear efficiency data

Survey gear efficiency for towed trawls and dredges is the probability of capture for individuals anywhere in the water column or sediments along the path swept by the trawl. Ideally, the area surveyed and the distribution of the stock coincides so that:

$$I_{k,y} = q_k B_{k,y}$$

$$q_k = \frac{a_k e_k u_k}{A}$$

$$e_k = \frac{A q_k}{a_k u_k}$$

$$K_t = \frac{A}{a_k u_k}$$

$$e_k = K_t q_t$$

Where $I_{k,y}$ is a survey observation in units equivalent to biomass (or numerical) density (e.g. kg per standard tow), $B_{k,y}$ is the biomass (or abundance) available to the survey, A is the area of the stock, a_k is the area swept during one tow, $0<e_k\leq 1$ is efficiency of the survey gear, and u_k is a constant that adjusts for different units.

Efficiency estimates from studies outside the CASA model may be used as prior information in CASA. The user supplies the mean and CV for the prior estimate of efficiency, along with estimates of A_k, a_k and u_k. At each iteration if the model, the gear efficiency implied by the current estimate of q_k is computed. The model then calculates the NLL of the implied efficiency estimate assuming it was sampled from a unimodal beta distribution with the user-specified mean and CV.

If efficiency estimates are used as prior information (if the likelihood weight $\lambda > 0$), then it is very important to make sure that units and values for the survey data (I), biomass or abundance (B), stock area (A), area per tow (a), and adjustments for units (u) are correct (see Example 1). The units for biomass are generally the same as the units for catch data. In some cases, incorrect specifications will lead to implied efficiency estimates that are ≤ 0 or • 1 which have zero probability based on a standard beta distribution used in the prior. The program will terminate if $e \leq 0$. If e • 1 during an iteration, then e is set to a value slightly less than one and a penalty is added to the objective function. In some cases, incorrect specifications will generate a cryptic error that may have a substantial impact on estimates.

Implied efficiency estimates are useful as a model diagnostic even if very little prior information is available because some model fits may imply unrealistic levels of implied efficiency. The trick is to down weight the prior information (e.g. $\lambda=1e^{-6}$) so that the implied efficiency estimate has very little effect on model results as long as $0 < e < 1$. Depending on the situation, model runs with e near a bound indicate that estimates may be implausible. In addition, it may be useful to use a beta distribution for the prior that is nearly a uniform distribution by specifying a prior mean of 0.5 and variance slightly less than 1/12=0.083333.

Care should be taken in using prior information from field studies designed to estimate survey gear efficiency. Field studies usually estimate efficiency with respect to individuals on the same ground (e.g. by sampling the same grounds exhaustively or with two types of gear). It seems reasonable to use an independent efficiency estimate and the corresponding survey index to estimate abundance in the area surveyed. However, stock assessment models are usually applied to the entire stock, which is probably distributed over a larger area than the area covered by the survey. Thus the simple abundance calculation based on efficiency and the survey index will be biased low for the stock as a whole. In effect, efficiency estimates from field studies tend to be biased high as estimates of efficiency relative to the entire stock.

Maximum fishing mortality rate

Stock assessment models occasionally estimate absurdly high fishing mortality rates because abundance estimates are too small. The NLL component used to prevent this potential problem is:

$$L = \lambda \sum_{t=0}^{N} \left(d_t^2 + q^2 \right)$$

where:

$$d_t = \begin{vmatrix} Ft - \Phi \; if \; Ft > \Phi \\ 0 \; otherwise \end{vmatrix}$$

and

$$q_t = \begin{vmatrix} \ln(Ft / \Phi) \; if \; Ft > \Phi \\ 0 \; otherwise \end{vmatrix}$$

with the user-specified threshold value Φ set larger than the largest value of F_t that might possibly be expected (e.g. $\Phi=3$). The weighting factor λ is normally set to a large value (e.g. 1000).

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Appendix B12: Forecasting methodology (SAMS model).

Dvora Hart, Northeast Fisheries Science Center, Woods Hole, MA.

The model presented here is a version of the SAMS (Scallop Area Management Simulator) model used to project sea scallop abundance and landings as an aid to managers since 1999. Subareas were chosen to coincide with current management. In particular, Georges Bank was divided into four open areas (two portions of the South Channel, Northern Edge and Peak, and Southeast Part), the three access portions of the groundfish closures, and the three no access portions of these areas. The Mid-Atlantic was subdivided into six areas: Virginia Beach, Delmarva, the Elephant Trunk Closed Area, the Hudson Canyon South Access Area, New York Bight, and Long Island.

Methods

The model tracks population vectors $\mathbf{p}(i,t) = (p_1, p_2,..., p_n)$, where $p_j(i,t)$ represents the density of scallops in the jth size class in area i at time t. The model uses a difference equation approach, where time is partitioned into discrete time steps $t_1, t_2,\ldots,$ with a time step of length • $t = t_{k+1} - t_k$. The landings vector $\mathbf{h}(i,t_k)$ represents the catch at each size class in the ith region and kth time step. It is calculated as:

$$h(i,t_k) = [I - \exp(\Delta t H(i,t_k))]p(i,t_k),$$

where I is the identity matrix and H is a diagonal matrix whose jth diagonal entry h_{jj} is given by:

$$h_{jj} = 1/(1+\exp(s_0 - s_1 * s))$$

where s is the shell height of the mid-point of the size-class.

The landings $L(i,t_k)$ for the ith region and kth time step are calculated using the dot product of landings vector $\mathbf{h}(i,t_k)$ with the vector $\mathbf{m}(i)$ representing the vector of meat weights at shell height for the ith region:

$$L(i,t_k) = A_i\, \mathbf{h}(i,t_k) \bullet \mathbf{m}(i)/(\, w\, e_i)$$

where e_i represents the dredge efficiency in the ith region, and w is the tow path area of the survey dredge (estimated as 8/6076 nm^2).

Even in the areas not under special area management, fishing mortalities tend not to be spatially uniform due to the sessile nature of sea scallops (Hart 2001). Fishing mortalities in open areas were determined by a simple “fleet dynamics model” that estimates fishing mortalities in open areas based on area-specific exploitable biomasses, and so that the overall DAS or open-area F matches the target. Based on these ideas, the fishing mortality F_i in the ith region is modeled as:

$$\mathbf{F_i = k{*}f_i{*}B_i}$$

where B_i is the exploitable biomass in the *i*th region, f_i is an area-specific adjustment factor to take into account preferences for certain fishing grounds (due to lower costs, shorter steam times, ease of fishing, habitual preferences, etc.), and *k* is a constant adjusted so that the total DAS or fishing mortality meets its target. For these simulations, $f_i = 1$ for all areas.

Scallops of shell height less than a minimum size s_d are assumed to be discarded, and suffer a discard mortality rate of *d*. Discard mortality was estimated in NEFSC (2004) to be 20%. There is also evidence that some scallops not actually landed may suffer mortality due to incidental damage from the dredge. Let F_L be the landed fishing mortality rate and F_I be the rate of incidental mortality. For Georges Bank, which is a mix of sandy and hard bottom, we used $F_I = 0.2F_L$. For the Mid-Atlantic (almost all sand), we used $F_I = 0.1F_L$.

Growth in each subarea was specified by a growth transition matrix *G*, based on area-specific growth increment data. Recruitment was modeled stochastically, and was assumed to be log-normal in each subarea. The mean, variance and covariance of the recruitment in a subarea was set to be equal to that observed in the historical time-series between 1979-2008. New recruits enter the first size bin at each time step at a rate r_i depending on the subarea *i*, and stochastically on the year. These simulations assume that recruitment is a stationary process, i.e., no stock-recruitment relationship is assumed. This may underestimate recruitment in the Mid-Atlantic if the recent strong recruitment there are due to a stock-recuit relationship.

The population dynamics of the scallops in the present model can be summarized in the equation:

$$p(i,t_{k+1}) = \rho_i + G\exp(-M\Delta tH)\,p(i,t_k),$$

where •$_i$ is a random variable representing recruitment in the *i*th area. The model was run with 10 time steps per year. The population and harvest vectors are converted into biomass by using the shell-height meat-weight relationship:

$$W = \exp[a + b\ln(s)],$$

where *W* is the meat weight of a scallop of shell height *s*. For calculating biomass, the shell height of a size class was taken as its midpoint.

Commercial landing rates (LPUE, landed meat weight per day) were estimated using an empirical function based on the observed relationship between annual landing rates, expressed as number caught per day (NLPUE) and survey exploitable numbers per tow. At low biomass levels, NLPUE increases roughly linearly with survey abundance. However, at high abundance levels, the catch rate of the gear will exceed that which can be shucked by a seven-man crew. The is similar to the situation in predator/prey theory, where a predator's consumption rate is limited by the time required to handle and consume its prey (Holling 1959). The original Holling Type-II predator-prey model assumes that handling and foraging occur sequentially. It predicts that the per-capita predation rate *R* will be a function of prey abundance *N* according to a Monod functional response:

$$R = \frac{\alpha N}{\beta + N},$$

where • and • are constants. In the scallop fishery, however, some handling (shucking) can occur while foraging (fishing), though at a reduced rate because the captain and one or two crew members need to break off shucking to steer the vessel during towing and to handle the gear during haulback.

The fact that a considerable amount of handling can occur at the same time as foraging means that the functional response of a scallop vessel will saturate quicker than predicted by the above equation. To account for this, a modified Holling Type-II model was used, so that the landings (in numbers of scallops) per unit effort (DAS) L (the predation rate, i.e., NLPUE) will depend on scallop (prey) exploitable numbers N according to the formula:

$$L = \frac{\alpha N}{\sqrt{\beta^2 + N^2}}.$$

The parameters • and • to this model were fit to the observed fleet-wide LPUE vs. exploitable biomass relationship during the years 1994-2004 (previous years were not used because of the change from port interviews to logbook reporting). The number of scallops that can be shucked should be nearly independent of size provided that the scallops being shucked are smaller than about a 20 count. The time to shuck a large scallop will go up modestly with size. To model this, if the mean meat weight of the scallops caught, g, in an area is more than 20 g, the parameters • and • in the above equation are reduced by a factor $\sqrt{20/g}$. This means, for example, that a crew could shuck fewer 10 count scallops per hour than 20 count scallops in terms of numbers, but more in terms of weight.

An estimate of the fishing mortality imposed in an area by a single DAS of fishing in that area can be obtained from the formula $F_{\mathrm{DAS}} = L_a/N_a$, where L_a is the NLPUE in that area obtained as above, and N_a is the exploitable abundance (expressed as absolute numbers of scallops) in that area. This allows for conversion between units of DAS and fishing mortality.

Initial conditions for the population vector **p** *(i,t)* were estimated using the 2009 NMFS research vessel sea scallop survey, with dredge efficiency chosen so as to match the 2009 CASA biomass estimates. The initial conditions from the 2009 surveys were bootstrapped using the bootstrap model of Smith (1997), so that each simulation run had both its own stochastically determined bootstrapped initial conditions, as well as stochastic recruitment stream.

Appendix B13: Modifications to the NEFSC sea scallop survey database.

Larry Jacobson and Dvora Hart, Northeast Fisheries Science Center, Woods Hole, MA.

Two modifications were made to the scallop survey database for this assessment. The first modification accommodated a change in the survey vessel and survey dredge. Beginning in 2007, the NEFSC scallop survey was carried out using the *R/V Hugh Sharp* in place of the *R/V Albatross IV*, new survey protocols and a modified survey dredge. In the database, the catch in each tow can be adjusted to account for differences in tow distance and potential differences in survey dredge efficiency. Specifically, the adjusted catch in tow *t* for surveys during 2008-2009 is $C_t^* = \phi C_t$ where C_t was the original catch and ϕ is the adjustment factor that converts survey catches during 2008-2009 surveys to *R/V Albatross IV* equivalent units. Variances for adjusted strata means were computed using Goodman's (1960) exact formula for the variance of the product of two random variables. Based on experimental work described in this assessment, ϕ=1/1.05=0.9524 to accommodate a 5% increase in tow distance for the new research vessel. For lack of information, the CV for the adjustment was assumed to be zero.

The second modification made it possible to compute survey abundance and biomass trends for GBK sea scallops back to 1979 instead of 1982. The years 1979-1982 were not used for GBK in the previous assessment because survey strata 6610, 6621, 6631, 6651, 6661, 6710, 6720 and 6740 were usually not sampled. In this assessment, Canadian data were used to fill these holes and Canadian data for other GBK strata were included as well (Figure 1). The Canadian survey also uses an 8' New Bedford style dredge with a liner. However the Canadians survey has a shorter tow distance (0.667 nm vs. 0.875 nm) and stratification is based on commercial LPUE in the preceding season rather than NEFSC shellfish strata. The Canadian data were adjusted for differences in tow distance based on the ratio of tow distances $C_s^* = \frac{0.667}{0.875} C_s$. Serchuk and Wigley (1986) showed that Canadian and US data from the same strata are similar after adjustment for differences in tow distance. Differences in stratification were therefore ignored. Canadian data were also used in the statistical model used to fill holes (strata not sampled in some survey years). Imputation procedures are described in NEFSC (2007).

References

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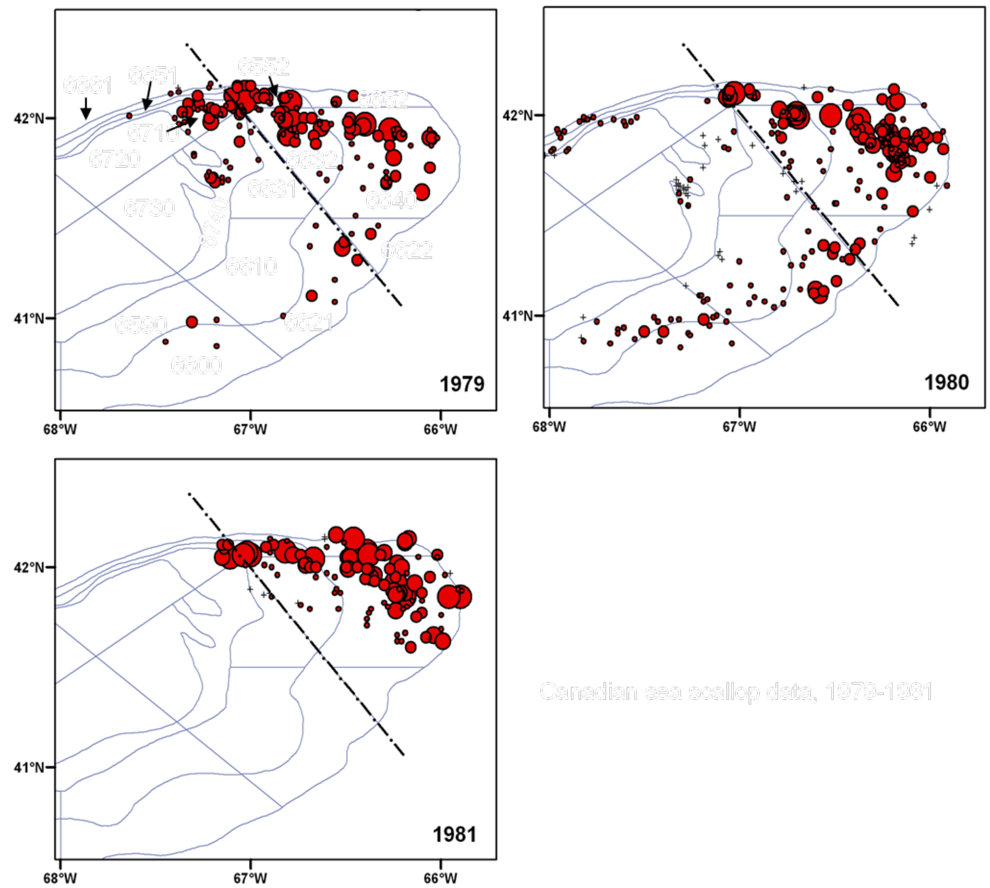
Northeast Fisheries Science Center. 2007. NEFSC. 2007. 45th Northeast Regional Stock Assessment Workshop (45th SAW): 45th SAW assessment report. Northeast Fish. Sci. Cent. Ref. Doc. 07-16.



Appendix B13-Figure 1. Location of Canadian sea scallop survey data for 1979-1981, which were used in this assessment. The size of the symbol in each plot indicates relative catch size.

Appendix B14: Comparison of surveys in the Nantucket Lightship Access Area during 2009.

Dvora Hart, Northeast Fisheries Science Center, Woods Hole, MA.

In 2009, three projects were funded by the sea scallop research set-aside program to intensively survey the Nantucket Lightship Access Area. One goal was to allow an effective comparison of density and shell height composition estimates. The three surveys were conducted by the Virginia Institute of Marine Science (VIMS), SMAST, and the HabCam team. The NEFSC lined dredge and SMAST drop camera “broad-scale” surveys, which are routinely carried out over the entire stock area, also covered the Nantucket Lightship Access Area, albeit less intensely. This analysis compares size-frequencies and abundance estimates from each survey.

Methods

The VIMS survey used two dredges towed side by side: a lined (38 mm) survey dredge (which is also used on the NEFSC survey) and a commercial dredge with 4” rings. The SMAST survey used the drop camera system used on their broad-scale survey including the primary “large” and secondary “small” cameras. The small camera gives better resolution because it is closer to the sea floor but covers less area (~0.8 sqm/drop). The HabCam survey used a towed digital camera system, towed at ~5 kts, taking overlapping digital images, each covering about 1 m^2 and with overlap between adjacent frames (Appendix B9). Table 1 gives more details on each survey.

The Nantucket Lightship Access Area was closed to scallop fishing in December 1994. It was reopened to fishing during portions of 2000 and 2004-2008. Previous surveys have observed three recent strong year classes: 1999, 2001, and 2004. The 1999 and 2001 year classes have been heavily fished. The remaining scallops from these year classes were expected to be around 150 mm shell height in 2009 (near their asymptotic size). The 2004 year class was lightly fished in 2008 only, and would be expected to be around 120+ mm shell height. All surveys were conducted in late spring or early summer in 2009, when the area was closed to fishing.

Results

Estimated shell height size-frequency (> 40 mm SH) from each survey were normalized to sum to one prior to the analysis. The VIMS survey dredge catches are used as a baseline for the size-frequencies analysis because the survey dredge is an important standard and shell height data collected by dredge surveys are relatively accurate (Jacobson et al. 2010).

The VIMS survey dredge showed the expected year class peaks at 120 and 150 mm SH, plus an incoming recruitment peak at 50 mm SH (Figure 1). The commercial dredge showed a similar size distribution for large scallops, but had reduced catchability for scallops less than 100 mm SH.

HabCam shell-height distributions were wider than the survey dredge shell height composition, probably due to less precise shell height measurements from photographs (Jacobson et al. 2010). Nonetheless, HabCam and the survey dredge are in reasonable agreement with no indication of dredge size-selectivity. The HabCam survey was conducted before the

VIMS survey, and the difference in timing may explain the differences between HabCam and VIMS in shell height distributions for smaller scallops that grow quickly.

The large drop camera survey suggests there is a much higher fraction of scallops in the 70-90 mm range than either the survey dredge or HabCam. The large camera size-frequencies are relatively noisy, with some evidence of reduced size-selectivity for small scallops. The divergence between the surveys may be due to the low sample size of the drop camera (315 scallops measured) and imprecision in shell height measurements (Jacobson et al. 2010). The small camera is intended to allow full detectability of small scallops, and indeed a higher proportion of small scallops were detected than with the large camera. However, the small camera data are noisier that the large camera data, due to the small number of scallops measured (76).

The NEFSC broad-scale survey had only 14 tows in the area. It found similar modes as the VIMS survey dredge, but in different proportions, likely due to the small sample size. The SMAST broad-scale large camera survey had a noisy shell height distribution, likely because of the small number of scallops measured (87).

Estimates of abundances are compared in Table 2. The dredge surveys were assumed to have an efficiency of 0.44 (see Appendix B4), whereas the optical surveys were assumed to have an efficiency of one. The individual 95% confidence intervals for each survey contain the inverse-variance weighted mean calculated for the abundance estimates from all of the surveys (205 million scallops). The three intensive dedicated surveys all had lower coefficients of determination (CV) than the broad-scale surveys.

Discussion and Conclusions

This study demonstrates the utility of fine-scale surveys for rotational area management in areas of relatively small size. Both abundance and the shell height composition data from the broad scale surveys are too imprecise because of the small sample sizes. It appears that the VIMS survey dredge gave the best estimate of shell height composition, as was assumed in the analysis. Both optical surveys showed evidence of shell height measurement errors. The SMAST survey did not measure sufficient scallops to estimate size-frequencies precisely. On the other hand, the optical surveys (SMAST and HabCam) had the lowest CVs for abundance. The HabCam survey had a remarkably low CV, due to its large sample sizes. Optical and dredge sampling have complementary attributes, and the ideal survey would probably include both types of sampling.

References

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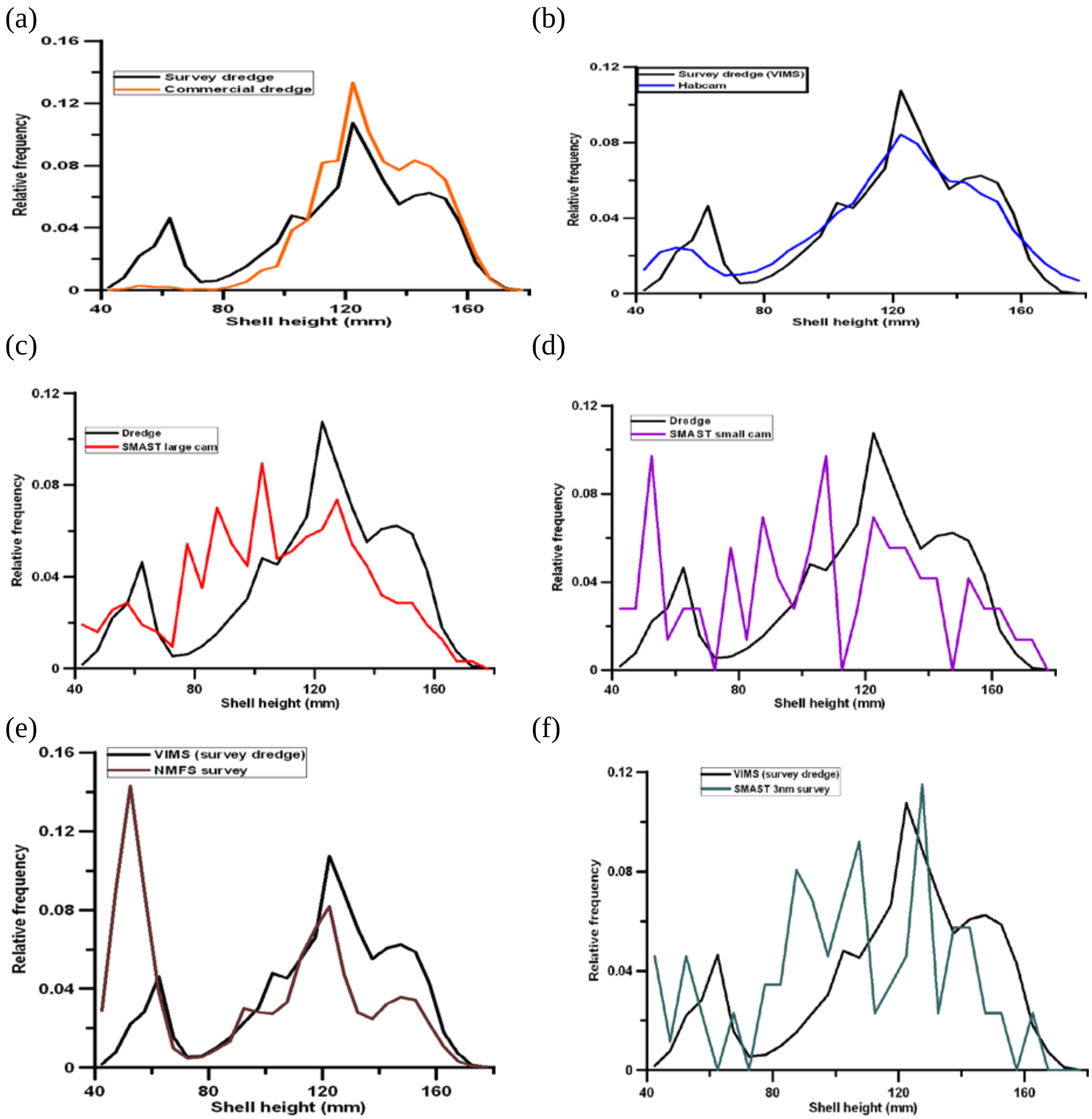
Appendix B14-Table 1. Basic characteristics of the surveys.

Survey	Gear	Design	Number of stations	Area swept (m^2)	Sea days	Number of scallops measured	Post-processing resources required
VIMS	Survey dredge	Systematic grid	91	409,500	4	13149	Low
VIMS	Commercial dredge	Systematic grid	91	767,813	4	16300	Low
SMAST	Large drop video camera	Systematic grid	164	1,940	2	315	Moderate
SMAST	Small drop video camera	Systematic grid	164	510	2	76	Moderate
Habcam	Towed digital still camera	Continuous transect	N/A*	123,500**	3	13644	High

*1.235 million images were collected, of which 1/10th were processed
**Processed images only

Appendix B14-Table 2. Abundance and biomass estimates from the surveys

Survey	Method	Assumed efficiency	Estimated abundance (millions)	CV	95% CI (millions)	Mean meat weight (g)	Estimated biomass (mt)
VIMS	survey dredge	0.44	259	0.14	192 to 334	34.0	10752
SMAST	large drop camera	1	240	0.13	183 to 305	25.0	5991
SMAST	small drop camera	1	234	0.16	166 to 313	24.6	5749
Habcam	towed camera	1	198	0.04	182 to 214	32.9	6782
NMFS broad-scale	survey dredge	0.44	100	0.45	32 to 206	32.5	3965
SMAST broad-scale	large drop camera	1	241	0.24	141 to 367	24.5	5902
Grand mean (inverse-variance weighted)		NA	207	0.035	193 to 231	34	7038
Broad-scale combo mean (inverse-variance weighted, NMFS and SMAST broad-scale surveys only)		NA	178	0.22	110 to 263	32.5	5798



Appendix B14-Figure 1. Plots of observed normalized shell heights for each survey. The VIMS survey dredge size-frequencies (black line) are included for reference on each plot. (a) VIMS commercial dredge. (b) HabCam. (c) SMAST large camera. (d) SMAST small camera. (e) Lined survey dredge. (f) SMAST broad-scale large camera survey. The NEFSC broad-scale survey data are not shown.

C. STOCK ASSESSMENT OF POLLOCK IN US WATERS FOR 2010

By: Northern Demersal Working Group (see Introduction for participant list)

Executive Summary

Terms of Reference:

1. Characterize the commercial and recreational catch including landings, effort, LPUE and discards. Describe the uncertainty in these sources of data, including consideration of stock definition.
2. Characterize the survey data that are being used in the assessment (e.g., regional indices of abundance, recruitment, state surveys, age-length data, etc.). Describe the uncertainty in these sources of data, including consideration of stock definition.
3. Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series, and characterize the uncertainty of those estimates.
4. Update or redefine biological reference points (BRPs; estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, and F_{MSY}; and estimates of their uncertainty). Comment on the scientific adequacy of existing and redefined BRPs.
5. Evaluate stock status with respect to the existing BRPs, as well as with respect to updated or redefined BRPs (from TOR 4).
6. Evaluate pollock diet composition data and its implications for population level consumption by pollock.
7. Develop and apply analytical approaches and data that can be used for conducting single and multi-year stock projections and for computing candidate ABCs (Acceptable Biological Catch).
    a. Provide numerical short-term projections (through 2017). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. In carrying out projections, consider a range of assumptions to examine important sources of uncertainty in the assessment.
    b. Comment on which projections seem most realistic, taking into consideration uncertainties in the assessment.
    c. For a range of candidate ABC scenarios, compute probabilities of rebuilding the stock by 2017.
    d. Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC.
8. Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in recent SARC reviewed assessments and review panel reports. Identify new research recommendations.

A new assessment model (ASAP, Legault and Restrepo 1998) is accepted as the best model for determining stock status for pollock (*Pollachius virens*). The base model for pollock estimates that spawning stock biomass in 2009 (SSB_{2009}) is 196,000 mt and the average fishing mortality on ages 5-7 ($F_{5\text{-}7}$) is 0.07. The criteria for determining stock status are based on reference points that use F40% as a proxy for FMSY, with SSBMSY calculated from projections at $F_{40\%}$. The overfishing criterion, calculated as the average F on ages 5-7, is $F_{40\%(5\text{-}7)}$=0.25 (this corresponds to a fully selected F of 0.41). The proxy for SSB_{MSY}, the B_{TARGET}, is estimated at 91,000 mt, with 5th and 95th percentiles spanning 71,000 to 118,000 mt. One half of SSB_{MSY} is the $B_{THRESHOLD}$ (45,500 mt). Comparing the current 2009 estimates of SSB and F to the MSY reference points, the stock is not overfished and overfishing is not occurring.

If the previous assessment model (AIM) had been used, the stock status would have been overfished with overfishing occurring. The new assessment model (ASAP) incorporates age structure and age-related biological processes, additional survey indices and their estimated variances, time-varying selectivity, commercial discards, and recreational landings and discards. The age-specific selectivities, and their evolution through time, are an important improvement. The fishery at the beginning of the time series exploited young, immature pollock, whereas the current fishery primarily exploits larger, mature fish. For all of these reasons, it is recommended that the previous assessment model, AIM, not be used for the current or for future assessments of pollock.

Previous assessments of pollock assumed a variety of stock definitions. Recent assessments of pollock in US waters are for "the portion of the unit stock of pollock primarily within the USA EEZ (NAFO Subareas 5&6) including a portion of eastern Georges Bank (Subdivision 5Zc) that is under Canadian management jurisdiction" (Mayo and Terceiro 2005). Canadian stock assessments treat the management unit within the Canadian EEZ separately (NEFSC 2002a). A review of information on population structure of pollock off the northeast US supports several alternative hypotheses of stock definition. Given uncertainties in stock structure and the considerable management implications, the Working Group developed a slightly refined stock definition that reflects the US jurisdictional unit (catch and survey information from current US waters).

Prior to 2000, pollock were assessed using virtual population analysis (VPA; e.g., Clark et al. 1981; Mayo and Clark 1984; Mayo and Figuerido 1993). Since 2000, pollock have been assessed using an index-based approach (Mayo 2001). The index approach was not designed for sophisticated projections, and performed poorly in recent projections to determine annual catch limits. For this benchmark assessment, an age-based approach to assessing pollock was attempted by updating fishery and survey catch-at-age and applying an Age-Structured Assessment Program (ASAP, Legault and Restrepo 1998). The revised stock definition, and transition to an age-based assessment, required a revision of the overfishing definition. Similar to most other groundfish managed under the Northeast Multispecies Fishery Management Plan (NEFSC 2002a), F_{MSY} is approximated as the fishing mortality that is expected to conserve 40% of maximum spawning potential ($F_{40\%}$, Clark 1991, 1993).

The role of pollock in the ecosystem was assessed using diet data. Estimates of pollock abundance were used to model pollock consumption. Results suggest that small pollock consume small invertebrates, primarily Euphausids, and large pollock prey on a mix of fish and invertebrates. Pollock is an ecologically important piscivore, but does not appear to be a dominant piscivore. Pollock is not a major prey species for any predator species.

Further research is needed to experimentally determine size-based selectivity of fishing gears, determine assessment and management units that most accurately reflect biological population structure, explore alternative survey techniques for off-bottom and hard-bottom habitats, and evaluate quality of age determination of old fish. The selectivity is especially important to resolve, as the ASAP model with dome-shaped survey and fishery selectivity implies the existence of a large biomass (35 – 70% of total) of pollock (i.e. cryptic biomass) that neither current surveys nor the fishery can confirm. Assuming full survey selectivity for ages 6 and above reduces stock biomass and associated biomass reference points by 20 – 50%. Notwithstanding this, the stock did not appear to be overfished in either case. Under the full selectivity assumption, long-term catches can be expected to be reduced by approximately 30%.

Introduction

Northern Demersal Working Group Meetings

Three meetings were held in preparation of the 2010 pollock assessment:

1. *Meeting with Pollock Fishermen* - January 22 2010 – MADMF Annisquam River Marine Fisheries Field Station, Gloucester MA (Appendix C1 includes a summary of the discussions). Participants included commercial fishermen (Terry Alexander, Richard Burgess, Matt Carter, Bill Gerencer, Bert Jongerden, Tom Kelley, Stephanie Neto, Jackie O'Dell, Frank Patania, Maggie Raymond, Mike Russo, Arthur Sawyer, Mike Walsh) and staff from the Northeast Fisheries Science Center (Liz Brooks, Steve Cadrin, Eric Thunberg) and the New England Fishery Management Council (Anne Hawkins, Tom Nies). A summary of the discussions is in Appendix C1.
2. *Data Meeting* - February 22-23 2010, NEFSC Woods Hole MA. Participants included Steve Cadrin (chair), Liz Brooks (lead assessment scientist), rapporteurs (Jessica Blaylock, Dan Goethel, Anne Hawkins, Kathy Sosebee, Susan Wigley) and others (Larry Alade, Russ Brown, Jon Deroba, Bill Duffy, Bill Gerencer, Jon Hare, Michael Jones, Richard Merrick, Tim Miller, Tom Nies, Paul Nitschke, Jackie O'Dell, Mike Palmer, Rebecca Rademeyer, Paul Rago, Dave Richardson, Fred Serchuk, Michelle Traver).
3. *Model Meeting* – March 29-April 2 2010, NEFSC Woods Hole MA. Participants included Steve Cadrin (chair), Liz Brooks (lead assessment scientist), rapporteurs (Jessica Blaylock, Bill Duffy, Dan Goethel, Anne Hawkins, Tom Nies, Julie Nyeland, Gary Shepherd) and others (Doug Butterworth, Rebecca Rademeyer, Richie Canastra, Laurel Col, Bret Elger, Jon Deroba, Jon Hare, Joe Idoine, Robert Gamble, Bill Gerencer, Michael Jones, Chris Legault, Jason Link, Rich McBride, Tim Miller, Paul Nitschke, Loretta O'Brien, Jim Odlin, Mike Palmer, Paul Rago, Maggie Raymond, Dave Richardson, Mike Russo, Brian Smith, Mark Terceiro). The group met by correspondence after the meeting, including a WebEx meeting on April 30 2010 to review the report and updated analyses with the full set of available data.

This Working Group (WG) report includes products from all three meetings and contributions from all participants.

Biology

Pollock are abundant on the western Scotian Shelf and in the Gulf of Maine (Mayo 1998; Figure C1). A major spawning area exists in the western Gulf of Maine and on Georges Bank, and several areas have been identified on the Scotian Shelf (Mayo et al. 1989a, Cargnelli et al. 1999). Spawning occurs from November through February with a peak in December (Collette and Klein Mac-Phee 2002). Juvenile pollock are common in inshore areas, but move offshore as they grow older. More than 50% of pollock are sexually mature by age 4 and maturation is essentially complete by age 6 (Mayo et al. 1989b). Pollock grow to a maximum length of 110 cm and maximum weight of 16 kg (Mayo 1998).

Fishery Regulations

A brief overview of New England groundfish management from 1977 to the present is provided as contextual information to help interpret fishery patterns and model results. The modern period of groundfish management began with implementation of the Magnuson-Stevens Act (M-S Act) in 1977. Since that time, all fishing for groundfish stocks within the U.S. Exclusive Economic Zone has been by U.S. vessels – no foreign fishing has been allowed. The management history can be broadly divided into four periods prior to 2010. Note that this discussion gives a broad overview. There were numerous other restrictions on gear, fishing practices, possession limits, etc. during all of these periods. Table C1 summarizes major elements of the federal groundfish management program since 1977.

1977–1981 - The first management plan used hard quotas for cod, haddock, and yellowtail flounder. There were various trip limits for these species. Catches of other groundfish stocks were not directly controlled. The fishery was open access – there were no limits on the number of permits. Minimum mesh size and minimum fish size regulations were also adopted, and seasonal closures to protect spawning fish were used.

1982–1993 - The quota system was abandoned in mid-1981 and replaced by a system that relied on technical measures (minimum mesh requirements, minimum legal sizes, etc.) and seasonal closures to protect spawning fish. There were complicated programs that allowed using mesh smaller than the minimum size to target other species. The fishery continued to be an open access fishery. Over time, the number of stocks subject to the plan increased. Mortality targets based on spawning potential were adopted.

1994–2003 - In response to stock declines and widespread overfishing, the number of permits was limited and a system of limiting fishing opportunities in the form of days-at-sea (DAS) was phased in over several years (Amendments 5 and 7). The DAS allocations did not constrain all permits and DAS use actually increased until 2001 (see Figure C2). DAS allocations remained unchanged from 1997 through 2001, but were reduced by a court order in 2002. The effort control system became more complex and used trip limits, seasonal and year-round closures, mesh size changes, and gear requirements. Various "exempted fisheries" were developed to facilitate targeting non-groundfish stocks. "Target TACS" (TTACs) for five stocks were adopted as a metric to evaluate the effectiveness of management measures, but exceeding these targets did not result in closing the fishery. The system for reporting catches was also completely revised in 1994 with the adoption of Amendment 5.

2004–2009 - Formal rebuilding programs were adopted that met requirements of the M-S Act. The DAS allocations were reduced in 2004, 2006, and 2009 (Amendment 13 and Framework 42). DAS were also categorized (identified as A, B, and C) with restrictions on each. Category A DAS could be used to target any stock; Category B DAS could only be used in certain programs designed to target healthy stocks, and Category C DAS could not be used but indicated a potential for future access. Several programs called SAPs (Special Access Programs) allowed targeting healthy stocks (primarily GB haddock) and the use of Category B DAS. Leasing of DAS between permits was adopted, which facilitated the transfer of fishing opportunities between permits. "Hard" (as opposed to target) quotas were adopted for a few programs and a few management units (GB yellowtail flounder was the only stock with a hard quota for all fishing).

A fifth period is expected to begin in 2010 with the expansion of a catch share program that will result in most of the fishery being subject to hard quotas. A key component is the formation of voluntary, self-selecting organizations identified as "sectors."

The WG identified regulations that were expected to affect fishery selectivity. Potential changes in selectivity might be anticipated after increases in minimum mesh sizes (1982-1983, 1994 and 1998) and after increases in minimum legal size of pollock (1986 to 1989). The working group agreed that changes in management regulations would be one consideration in the development of the assessment model, and specifically in the determination of blocks of years when selectivity could be assumed constant.

Assessment History

The first analytical stock assessment completed for the Gulf of Maine, Georges Bank and Scotian Shelf (ICNAF areas 5 and 4VWX) was in 1976. Results from catch curves indicated that fishing mortality in the 1970s exceeded the level associated with maximum yield-per-recruit (ICNAF 1976). After the international boundary was defined in 1984, Canada assessed pollock on the Scotian Shelf (4VWX) separately, but the US continued to assess pollock in 4VWX and 5. The Scotian Shelf, Georges Bank and Gulf of Maine stock was assessed using virtual population analysis beginning in 1981 and continuing through the mid-1990s (Clark et al. 1982; Mayo and Clark 1984; Mayo et al. 1989b, Mayo and Figuerido 1993, Mayo 1998). Spawning stock biomass had been declining since the mid-1980s, and fishing mortality was estimated to be 0.72 for ages 6+ in 1992, above $F_{20\%}$=0.65 (Mayo and Figuerido 1993).

The analytical assessment was replaced with an index-based assessment (Mayo 2001) that used total commercial landings in NAFO areas 4VWX, 5, and 6, and the NEFSC fall survey. Recent assessments of pollock in US waters are for "the portion of the unit stock of pollock primarily within the USA EEZ (NAFO Subareas 5 and 6) including a portion of eastern Georges Bank (Subdivision 5Zc) that is under Canadian management jurisdiction" (NEFSC 2002b). The overfishing criterion was defined as the relative exploitation rate that allowed replacement, and the overfished criterion was based on the general magnitude of NEFSC fall survey biomass index from the 1980s (NEFSC 2002b). In 2001 and 2005, the index assessment determined that the stock was not overfished, and overfishing was not occurring (NEFSC 2002a, Mayo and Terceiro 2005). In 2006-2007, the fall survey index decreased, and the 2008 index-based assessment determined that the stock was overfished and overfishing was occurring (NEFSC 2008). The index-based assessment was updated with 2008 catch and survey data, but results were rejected as a basis for catch advice in 2009 (Multispecies Plan Development Team and New England Scientific and Statistical Committee 2009).

Stock Definition

Geographic Variation –

Mayo et al. (1989a, 1989b) found no significant differences in allozyme frequencies between fish in US and Canadian waters, but allozyme differences among coastal and marine populations are rare, even for many populations that are now considered to be reproductively isolated according to more sensitive genetic markers.

Two studies found morphological differences between western Scotian Shelf and Georges Bank-Gulf of Maine. McGlade (1983) concluded that meristics were significantly different between areas 5 and 4X. McGlade and Boulding (1986) also reported differences between areas 5 and 4X using morphometrics. Growth rates on the Scotian Shelf were different between pollock in 4X and 4VW Neilson et al. (2006), but growth of pollock in US and Canadian waters has not been compared.

Geographic Distribution and Patterns of Abundance –

Larval distributions indicate three relatively discrete spawning areas: 1) in the Gulf of Maine, 2) on the western Scotian Shelf, and 3) on the eastern Scotian Shelf (Figure C3; from Richardson & Hare WG presentation). Pollock larvae were rarely found in samples over the deep waters of the Gulf of Maine indicating limited mixing during early life stages of fish from US and Canadian waters.

NEFSC trawl surveys indicate a generally continuous distribution of pollock across the Gulf of Maine and western Scotian shelf (Figure C4). This indicates that it is likely that mixing occurs during adult life stages, although the rate of mixing cannot be determined. Despite large inter-annual variations in survey indices, abundance trends from NEFSC and DFO surveys generally agree. All show a general pattern of high abundance early in the time series, declines during the middle period (early and mid 1980s), with some increases in recent years. There is more divergence among surveys in recent years.

Much of the catch from US waters appears to be from the western and central Gulf of Maine, with some landings near the US/Canadian boundary of Georges Bank (see section on fishing effort). These landings are probably a mixture of fish spawned in both 4X and 5. Canadian landings trends appear to differ between the Eastern and Western Scotian Shelf components (between 4X and 4VW).

Tagging –

Three main tagging studies have been carried out for Pollock in US waters. An historical study was undertaken by Schroeder from 1923-1927. While only a subset of this data has been examined to date, a preliminary evaluation of the data found less than 100 recaptures from nearly 3800 releases. The data from the Schroeder study was hand written in journals with locations generally specified by landmark; thus, both the release and recovery locations are fairly imprecise, although the general direction of movement can be inferred and some mixing is suggested between US waters and the Scotian Shelf (Figure C5). More recent studies were carried out by Clay et al. (1989) and Neilson et al. (2003, 2006). The general pattern of release and recovery locations indicated relatively high connectivity (~16%) between fish tagged on the western Bay of Fundy (4Xs) and recaptured in the western Gulf of Maine. This is in contrast to fish tagged on the eastern Bay of Fundy (4Xr), which had very few recoveries in US Waters (~4%, primarily the northeast edge of Georges Bank). The tagging took place between 1978-1984, with recoveries from 1979-1990. Both Neilson et al. (2006) and Steele (1963) suggest a population of fish in the western Bay of Fundy that migrate for spawning purposes to the southern Gulf of Maine (Figures C6a and C6b). Neilson (2006) suggests that this is a small fraction of the overall western Canadian pollock stock. Mixing between 4X and 4VW was less frequent, and mixing of pollock in 4VW and those in 5 is limited. Tagging data suggests that pollock in the US and on the Western Scotian Shelf could be considered a unit stock based on historical estimates of movement, however, the fish on the eastern Scotian Shelf appear to be a separate stock unit.

Multidisciplinary Studies –

Neilson et al. (2006) synthesized much of the data available on pollock stock structure and concluded that there was enough evidence to suggest that three stocks existed: 1) western

Gulf of Maine coastal population; 2) western Scotian Shelf and Bay of Fundy and 3) eastern Scotian Shelf.

The WG concluded that pollock within US waters should be treated as a single stock (i.e. areas 5 and 6 were the same stock), because the majority of fish appeared to be located in the Gulf of Maine, with some fish and landings on Georges Bank and few pollock south and west of the Great South Channel. The more difficult decision was to determine the relationship between US and Scotian Shelf stocks. The objectives of stock assessment and fishery management were also considered by the WG. For management purposes, assessment of pollock in US waters would be ideal, if the population dynamics of pollock in US waters is not influenced by connectivity with the Scotian Shelf. For the purposes of stock assessment, population dynamics should be primarily influenced by processes within the stock area, all catch from the assessment unit should be accounted for, and all survey data should be representative of the stock.

Scientific information on population structure of pollock off New England provides equivocal evidence for three possible hypotheses about the appropriate assessment unit:

1. *US portion of NAFO areas 5 and 6 (Gulf of Maine and Georges Bank)* – This is the assessment unit evaluated by the 2008 assessment (GARM III). Assessment of pollock in areas 5 and 6 is supported by larval distributions, morphology and recent survey trends. Larval distribution suggests that spawning in the area from southwest Gulf of Maine to Georges Bank is distinct from another spawning area on the western Scotian Shelf (MARMAP data presented by D. Richardson and J. Hare). Morphometry is significantly different between the western Gulf of Maine and the Scotian Shelf (McGlade and Boulding 1986). Recent trends in surveys of the western Scotian Shelf and in areas 5 and 6 provide different perspectives of stock development. A recent multidisciplinary review of stock structure that was focused on the Canadian maritimes (Nielsen et al. 2006) concluded that there are three stocks of pollock in the area: 1) "the western Scotian Shelf (including the eastern Bay of)", 2) "on the eastern Scotian Shelf" and 3) "a coastal population in the western Gulf of Maine that overlaps into Canadian waters." From a practical perspective, a stock assessment based on catch and survey data in US waters would support evaluation of US catch limits without the need to forecast Canadian catch.
2. *NAFO areas 4Xo-s, 5 and 6 (Gulf of Maine, Georges Bank, and the western Scotian Shelf)* – Combined assessment of Georges Bank, the Gulf of Maine and the western Scotian Shelf is supported by tagging data, fishery distributions, long-term survey trends, and growth rates. Considerable movement of juveniles and adults among all three areas is documented by tagging data (Schroeder 1923-27, unpublished; Clay et al. 1989; Nielsen et al. 2006). Most recent US fishery catch is from the western Gulf of Maine, with a small amount of catch on NE Georges Bank adjacent to the international boundary. Unlike the divergent trends in recent survey indices, US and Canadian surveys both suggest a relatively abundant stock in the 1980s, depletion in the early 1990s, and rebuilding since the mid 1990s. Growth rates appear to be different between the eastern and western Scotian Shelf (Clay et al. 1989). Assessment of a transboundary resource would pose considerable uncertainty for fishery management with respect to management objectives, allocations and projected catch.
3. *NAFO areas 4VWX, 5 and 6 (Gulf of Maine, Georges Bank, and the Scotian Shelf)* – Combined assessment of the entire US and Scotian Shelf is supported by genetics, tagging and survey distributions. Analysis of allozymes suggests no genetic differences among these areas (Mayo et al. 1989a, 1989b). Tagging data suggest some connectivity

between US waters with the entire Scotian Shelf (Nielsen et al. 2006). Survey data suggests a continuous distribution of pollock along the Scotian Shelf. Assessment of pollock in NAFO areas 4VWX, 5 and 6 would be difficult, because no single survey covers the entire distribution of the resource and would complicate management, because Canada assesses and manages eastern and western Scotian Shelf as separate units.

Given uncertainties in stock structure and the considerable management implications, the Working Group decided to develop an assessment that reflects the US management unit (option 1 above, with US catch and survey information from survey strata that are in US waters: strata 13-30, 36-40). This U.S. management unit complements the Canadian management unit on the Scotian Shelf and Canadian portions of Georges Bank and the Gulf of Maine (Stone et al. 2009).

The Fishery

TOR 1: Commercial and Recreational Catch

Characterize the commercial and recreational catch including landings, effort, LPUE and discards. Describe the uncertainty in these sources of data, including consideration of stock definition.

Commercial Catch

Pollock were traditionally landed as bycatch in various demersal otter trawl fisheries, but directed otter trawl effort increased during the 1980s, peaking in 1986 and 1987 (Mayo 1998). Directed effort by US trawlers declined in the 1990s and early 2000’s, but there have been recent increases in landings that may reflect increased targeting of pollock. Similar trends have also occurred in the U.S. winter gillnet fishery.

U.S. commercial landings increased from approximately 4,000mt per year in the late 1960s to a peak of 24,000mt in 1986 (Figure C7, Table C2). Landings rapidly decreased to 4,000mt in 1996, and generally increased to 10,000mt in 2008. Historical landings were primarily from trawl fisheries, but contributions from gillnet fisheries generally increased, and the recent fishery landings are split 60%-40% between trawl and gillnet fisheries, respectively (Figure C7). Among the thirteen species managed by the Northeast Multispecies Fishery Management Plan, pollock was second only to cod in landed weight from 1996 through 2008. From 2006 to 2008, pollock landings were higher than those of any other groundfish in this multispecies fishery. Pollock is relatively low in value, however, with the annual average price never exceeding $1.00/per pound during this period. From 1996 to 2008 pollock ranked seventh in landed value. In recent years its revenue contribution increased with the increase in landings and it has ranked in the top five species for revenues since 2006.

Landings were mostly from unclassified market category until minimum legal size regulations were imposed in the late 1980s. At that point, the majority of landings were from the ‘large’ market category (Figure C8). In the last decade, landings from ‘medium’ and ‘small’ market categories went from being about equal to about 3:1 in favor of the ‘medium’ category. Landings by market category should be considered with caution because there is uncertainty regarding which lengths/weights were used as cull points throughout the time series. In particular, the ‘medium’ market category is primarily used in Portland, Maine, and it is unclear whether these fish would be have been classified as ‘small’ or ‘large’ had they been landed in a different port. Consequently, it might be more appropriate to consider landings by size composition (catch at age) only instead of market category. Historically, this was more of a winter fishery, with higher landings in quarters 1 and 4. More recently, landings have been approximately equally distributed among seasons (Figure C9).

Port samples of size and age structure are summarized in Table C3. Sampling intensity has been good since the early 1980s. Landed catch at age shows some relatively strong year-classes in the 1970s and 1980s (Figure C10). Age-based analyses begin in 1970, based on the availability of commercial catch at age data. At the data meeting, the working group decided that age-based analyses should attempt to model ages 1 to 12+, as had been done in earlier VPA analyses. The motivation for this decision was that pollock are fully mature by age 7, and even though they are still growing at age 12, the weight of the 12+ groups would be derived from empirical observations. This decision was revised at the model meeting to aggregate the data with a 9+ group.

Commercial discards (D) were estimated using the Standardized Bycatch Reporting Methodology (Wigley et al. 2007) in which the ratio of discarded pounds of pollock ($d_{pollock}$) to kept pounds of all species ($k_{all\_species}$) for each fleet is sampled by observers at sea, and the ratio is expanded to total pollock discards according to commercial landings of all species ($K_{all\_species}$) by fleet.

$$D = \frac{d_{pollock}}{k_{all-species}} K_{all\_species} \qquad \text{(C.1)}$$

Estimates of pollock discards were stratified by NAFO areas (5 and 6), gear (otter trawl and gillnet), and mesh (small, large, extra-large). Discards were estimated for years 1989 to 2008 (data were not available for 2009, so an assumed value equal to 2008 discards was used). The estimates of discards ranged from 1% to 8% of US commercial landings, with an average of 3% for all years estimated. The four fleets that account for nearly all pollock discards were small-mesh otter trawl, large-mesh otter trawl, large-mesh gillnet, and extra-large mesh gillnet (Table C4). Estimates of pollock discards from other fleets (longline, handline, small-mesh gillnet, scallop dredge and midwater trawls) were excluded from discard estimation because of periods with low sampling intensity and apparently low magnitude of pollock discards. Discards from the shrimp fishery were also considered to be negligible.

Discard estimates for small-mesh otter trawl in 1994 and 1997 were approximated using discard observations from adjacent years. Discards were assumed to be negligible before 1989, because estimated discards are a small portion of catch, there were few reasons to discard pollock before 1989, and there is no viable alternative for estimating historical discards. According to fishermen, there was no market for small pollock in some ports prior to the mid 1980s, which suggests that some discarding might have occurred on fish below a landable size prior to 1989. However, more extensive analysis based on landed and survey size distributions by port or survey strata would be needed to evaluate landed trends and to consider appropriate methods to hindcast historical discards.

Commercial Fishing Effort

Two data sources are available to provide information on the location of fishing effort: fishing vessel logbooks and fishery observer reports. Each vessel operator submits a Vessel Trip Report (VTR) at the end of each trip that includes position, fishing activity, and catch information. Reporting regulations require only that the VTR indicate the general area of fishing activity in a statistical area. While the regulations require submitting a separate VTR page for every statistical area fished, compliance with this requirement is uneven. VTR information thus provides an overview of reported general trip level fishing activity but does not provide precise fishing location information.

Observer reports provide detailed fishing information on a tow-by-tow (or haul-by-haul) basis, but not all trips are observed, and not all tows on every trip are observed. Levels of observer coverage in the groundfish fishery were generally low prior to 2000, but have increased in recent years. Changing priorities can modify the distribution of trips over time. As a result, drawing conclusions from observer data can be difficult because the observations are influenced not only by the distribution of fishing activity but by the allocation of observer resources. Observer data remains the best source of precise location information and detailed fishing activity.

The goals of these examinations were to: 1) determine if there is evidence in the geographic distribution of fishing activity to support identification of different stock or management units for pollock within the U.S. Exclusive Economic Zone; 2) determine if large pollock catches are associated with specific areas; and 3) determine if there is evidence of changes in the distribution of pollock catches.

VTR Database Analyses

Data –

The VTR database was queried to select all fishing trips that landed any pollock during the years 1996 through 2008 (the latest year for which complete VTR data was available). For each such trip, other data elements were retrieved including the year and month of landing, latitude and longitude where the haul began, gear code, days absent, trip ID and permit number. Data elements were not selected for other fields for this exercise.

To facilitate analysis the data was plotted using ArcGis© and maps were created showing the number of trips that caught pollock and the total weight of pollock caught for each year. Each subtrip was binned into a ten-minute square based on the reported location of the beginning of the haul. The ten-minute squares were color coded based on the difference between the average number of subtrips in a square and the value of the specific square. This difference is measured in standard deviation units from the mean number of subtrips in a square for each year.

Results –

The number of sub-trips in each ten-minute area per year that caught pollock is shown in Figure C11. The total weight of pollock caught in each ten-minute area per year is shown in Figure C12. A comparison of the two figures suggests that an increase in pollock landings is not necessarily closely associated with an increase in number of trips. Large pollock catches were reported in areas with few reported trips.

It appears that the range of pollock declined between 1996 and 2008, since the offshore areas that experienced high pollock trips in the early years seem to have fewer in 2004-2008. However, many fewer trips were reported in this area in 2004-2008 compared with the inshore area. It therefore does not necessarily follow that the range is contracting.

The analysis suggests that pollock are widely distributed in the deep water areas of the Gulf of Maine and Georges Bank. There seem to be areas with larger pollock catches (landings) relative to the number of trips taken further offshore. It is difficult to determine from these figures whether the presence of pollock is continuous in the Gulf of Maine and the northern side of Georges Bank, or whether there could be distinct areas with high concentrations.

Observer Database Analyses

Data –

The observer database was queried to select all trawl (negear=050) and sink gillnet (negear=100) tows from trips that landed any of the regulated groundfish species or monkfish during the years 1989 through 2009. A single record was created for each such tow that summarized total caught weight (in live weight) and the weight caught of the regulated groundfish species, monkfish, and skates. Other data elements retrieved were the year, quarter, and month of landing, position haul began, gear code, and target species. Data elements were not selected for gear characteristics, soak time, vessel size, or haul duration for this exercise.

The number of trawl tows selected by this query varied over time. From 1989 through 2000 the average number of tows that met the selection criteria was 1,713. The average increased to 4,208 during 2001-2003, and then tripled to 13,365 from 2004 through 2009. The peak year was 2005 (23,064 observed tows selected). The increases since 2002 are the result of increased funding for the observer program and are not related to an increase in fishing effort. On the contrary, groundfish fishing activity declined by over 50 percent from 2001 to 2009. Most of the analyses focus on the period since 2002 when there were increased levels of observer coverage.

The number of sink gillnet hauls observed over time was more consistent than was the case for trawl tows. From 1989 to 2000 the average number observed was 1,661, while from 2001 through 2009 it was 1,663. The peak year was 1991, with 4,175 observed hauls selected, while the low was 1989, with 348. From 1999 through 2002 the average was 607. These more consistent coverage levels are likely due to interested in observing sink gillnet activity to document marine mammal interactions. Because of the more consistent coverage, the sink gillnet analyses that follow will consider the 1992-1999 and 2002-2009 time periods.

To facilitate analysis the data was also plotted using ArcGis© and each tow was binned into a ten-minute square based on the location of the beginning of the haul. The number of squares with a tow gives a simple metric of the geographic extent of observer coverage in a year (but this metric is difficult to interpret because of changing observer coverage).

Trawl Results –

The number of ten-minute squares with an observed tow increases as the number of observed tows increases. Up to about 4,000 observed tows, the number of ten-minute squares increases rapidly in a linear fashion (R^2=0.81, with the slope significant $p<0.01$). The increase slows considerably above this number of observed tows but the slope remains significant. This suggests that there are only small increases in the geographic distribution of observed tows once observer effort is sufficient to observe over 4,000 – 6,000 trawl tows. A similar relationship holds for the number of ten-minute squares with an observed pollock tow below 4,000 observed tows; above 4,000 observed tows, there was a slower increase and the slope of the increase is marginally not significant ($p=0.055$). A similar relationship was noted between the number of observed tows and the number of ten-minute squares with an observed pollock tow. Additional analyses will focus on the period 2002 through 2009 since these years have more observations and there is less influence on the results from changes in levels of observer coverage.

It appears that the range of pollock declined between 2002 and 2009, because the number of squares with an observed pollock tow declined from 50 percent of the squares with an observed tow to 33 percent of the squares with an observed tow. However, this interpretation ignores that the distribution of observer coverage also changed: tows were observed in 317 ten-minute squares in 2002 and 546 in 2009. When squares with an observed tow in both years are considered (258), the number of tows with an observed pollock tow increased slightly from 134 in 2002 to 139 in 2009.

Pollock were observed in tows throughout the Gulf of Maine and the northern part of Georges Bank. Generally, where there are many observed tows, there are many observed tows with pollock. Only in the shallower areas of Georges Bank is there much difference between the location of observed tows and the location of observed pollock tows. Large pollock tows, however, are more localized. They tend to be located along the 50 and 100 fathom depth contours on the north side of Georges Bank and then extend north along the western edge of the western Gulf of Maine closed area (which is near the 100 fathom curve). The presence of pollock seems to be continuous in the Gulf of Maine and then northern side of Georges Bank, a fact that cannot be determined from the VTR data alone.

Two additional analyses were performed to identify areas with pollock concentrations. In the first, catches on all observed tows in each ten-minute square were combined and the total catch of pollock as a percentage of total observed catch in that square was determined (Figure C13). From 2007 through 2009 the number of squares where pollock catch was more than half the observed catch increased. The areas also seem relatively constant over time, primarily along the 100 fathom curve east of Cape Cod and the western Gulf of Maine closed area.

Sink Gillnet Results –

The number of ten-minute squares with an observed haul increases as the number of observed tows increases. As was the case with trawl observations, there seem to be two rates. Up to about 1,300 observed tows, the number of ten-minute squares increases rapidly in a linear fashion (R^2=0.91, with the slope significant p=0.00). Above this number of observed trips the slope of the regression is nearly flat but is not significant (p=0.142). Unlike trawl tows, the number of observed hauls with pollock does not seem related to the number of observed hauls.

Pollock were observed in hauls throughout the Gulf of Maine and the northern part of Georges Bank. When the location of observed sink gillnet hauls during 1992-1999 is compared to 2002-2009, one change is obvious. In the early 1990’s sink gillnet hauls were observed along the entire coast of Maine. Pollock were frequently caught in the coastal areas east of 69-30W longitude. There were large hauls observed along the 100 fathom curve as far east as the Hague Line that divides U.S. and Canadian waters. Beginning in 1994, there were dramatically fewer observed sink gillnet hauls in these eastern areas. There was a slight increase in 1995, but then there were almost no observed hauls in the area through the end of the first period, and then through the 2002-2009 period examined. Sink gillnet observed hauls in 2004 – 2009 that caught pollock were concentrated in the inshore Gulf of Maine area off Massachusetts, New Hampshire, and southern Maine and the 100 fathom curve in the central Gulf of Maine. Effort as indicated by observed sink gillnet hauls did not extend into the northeastern part of the Gulf of Maine where it was common in the early 1990’s.

Figure C14 shows pollock as a percent of observed sink gillnet catch from 2001-2009. There are few ten-minute squares where pollock was more than 25 percent of the observed catch. The instances where this does occur tend to be along the 100-fathom curve in the central Gulf of Maine. The obvious change in the distribution of observed sink gills after 1994/1995, as well as the change in the distribution of hauls catching pollock, warranted further investigation. The changes could reflect a shift in the distribution of pollock that is not evident from the trawl data because there are fewer observations in the early 1990's. The timing of the change, however, also suggests that it could be related to the adoption of a limited entry program in the fishery in 1994. The program is often criticized for not awarding permits to small boat fishermen from the coastal communities of eastern Maine.

To determine if the regulatory change may be responsible for the lack of observed sink gillnet trips off eastern Maine after 1994/1995, the landing port for trips that had observed hauls north of $43^{o}30'$N and east of $69^{o}30'$W was determined. During the 1989-1993 period before the regulatory change, almost all of the hauls were on trips that landed in coastal Maine ports by vessels that claimed a Maine homeport. The permit database was queried to determine whether these vessels received a limited access multispecies permit in 1994; most did not. The absence of observed sink gillnet hauls in this area after 1994/1995 can be attributed, at least in part, to the fact that vessels that fished with sink gillnets in the area in 1992 and 1993 did not receive a limited access permit when that program was adopted in 1994.

The VTR data indicate that pollock is caught by vessels widely distributed in the Gulf of Maine and Georges Bank. There are areas that produce larger pollock catches on a fairly consistent basis. The observer tow-by-tow data – both trawl tows and sink gillnet hauls - suggests pollock is continuously distributed throughout the area. The sink gillnet observed hauls seem to indicate that pollock is no longer caught in the inshore areas off the eastern coast of Maine. This may reflect the fact that vessels from Maine that fished in this area before 1994 did not receive limited access multispecies permits when Amendment 5 was implemented in 1994. It is also clear from the VTR and observer information that there has been little groundfish fishing activity inside the 100 fathom curve off eastern Maine in recent years. Because of varying levels of observer effort and numbers of reported VTR trips, this investigation did not draw conclusions on possible changes in the geographic distribution of fishing effort over time.

The WG concluded that CPUE trends have limitations due to changes in regulations over time (DAS, area closures, etc); however, trends in nominal effort (number of trips and/or number of days absent) might be useful for interpretation purposes only (not for use in model).

Recreational Catch

The time series of recreational catch is highly variable from year to year (Figure C15, Table C2). Recreational catch peaked at 1867mt in 2008, which is consistent with fishermen's accounts of encountering large numbers of pollock in that year. However, recreational catch of pollock decreased in 2009 to 896mt. Since 2001, the shore component decreased relative to the party/charter and private/rental components, with the private/rental component accounting for 50% or more of the recreational pollock catch. Recreational catch is small relative to commercial landings and has generally been 10% or less. However, from 2000-2004, recreational catch is estimated to have contributed 15-24% of total catch (commercial catch was near the lowest values in the time series for these same years, Table C2). There are no recreational catch estimates from the statistically designed sampling program (MRFSS) prior to 1981.

A tagging study (Clay et al. 1989) estimated 16% total mortality from a hook fishery in a three-month period, 11% of which was attributed to tagging of fish. That study suggested that neither 100% mortality nor 100% survival would be an obviously justifiable assumption for recreational discard mortality of pollock. In the absence of more information, the working group chose to assume 100% mortality of discarded recreational catch (B2). This assumption is also consistent with the 100% discard mortality assumed for commercial discards. Furthermore, because recreational catch is a minor component of the total catch, assuming 100% mortality was not expected to contribute undue influence on model results.

The WG decided that the length-frequency of discards would be best represented by samples of the recreational kept catch (A and B1). Recreational age samples are not available, so age compositions need to be borrowed from other data sources. The WG agreed that survey data would provide the most equivalent information to the recreational catch.

Estimates of recreational catch of pollock begin in 1981. The WG decided to assume negligible recreational catch prior to 1981, as there is no agreed method and scant data upon which to base hindcast estimates. Furthermore, the magnitude in recent years is a minor component of total catch, and it is assumed that any recreational catch prior to 1981 would not have exceeded the recent amounts.

Resource Surveys

Term of Reference #2: Survey Data

Characterize the survey data that are being used in the assessment (e.g., regional indices of abundance, recruitment, state surveys, age-length data, etc.). Describe the uncertainty in these sources of data, including consideration of stock definition.

Several surveys are available to provide indices of relative abundance. The properties of each survey were examined to determine whether it should be used for stock assessment of pollock. Table C5 provides a summary of survey attributes.

Given the stock definition described above, survey indices will be based on data from all strata that have been consistently sampled in US waters (NEFSC strata 13-30, 36-40; Figure C16). While several of these strata straddle the Hague Line, the working group decided that dropping those strata would create a larger discontinuity between the fishing area and the survey area, and would likely increase the estimated variance. Both the fall and spring surveys have large inter-annual variation (Figures C17 and C18). The NEFSC fall survey series generally corresponds with the exploitation history: the survey index declines from high biomass in the late 1970s to extremely low biomass in the mid 1990s, consistent with annual landings exceeding 20 000t during the same period; biomass increased in the late 1990s when landings were <6 000t; survey biomass decreased again as recent landings approached 10 000t. The spring survey does not correspond as well with the exploitation history.

Previous assessment models (VPA, AIM) dealt only with the annual index point estimate, with all points given the same weight in the objective function. In an attempt to avoid undue influence from some of the year effects, indices for those earlier models were derived from log-retransformed data (with a value of 1.0 added to observed zeros). For the present assessment, the new assessment model (ASAP) has the capability to apply index-specific weights as well as year-specific weights within each index. The working group decided to use the NEFSC spring and fall survey N/tow without transformation, and to use the annual estimates of coefficient of variation (CV) as annual weighting factors. No additional weights were applied to the indices.

Several changes to the fishing system occurred in the NEFSC spring and fall survey time series. In 1985, trawl doors were changed from ‘BMV oval’ doors to ‘Euronet Polyvalent’ doors. Calibration experiments for the two sets of survey doors included only nineteen paired tows that caught pollock. Conversion coefficients were significantly different than zero (p=0.03 for number, p=0.01 for weight), with a door coefficient of 2.21 (95% CI 1.11 - 4.30) for number per tow and 2.90 (95% CI 1.38 - 5.54) for weight per tow. Although most surveys were done by the R/V Albatross, the R/V Delaware was used intermittently. Vessel calibration experiments included 32 paired tows that caught pollock, and conversion coefficients were not significantly different than zero (P=0.92 for number, p=0.66 for weight). In 2009, the R/V Albatross was permanently replaced by the FSV Bigelow. Nineteen paired tows in the Albatross-Bigelow calibration experiment caught pollock (8 in spring, 11 in fall). A peer review panel offered general guidelines for calibration protocols:

- If there are less than 30 paired observations with positive catches, do not attempt any conversion.
- If there are less than 30 paired observations with positive catches in any one season, seasonal conversion are not appropriate.
- Pollock catches are too low to derive a reliable conversion factor, and the comparison is driven by one large value.

Given the low sample sizes and imprecise estimates from calibration, the WG decided that calibration coefficients will not be used to adjust survey data for changes to survey systems (e.g., doors, nets, vessels).

Several analyses were explored to investigate potential factors in survey catchability. In response to the observation that pollock distribution may have shifted to deeper habitats (Nye et al. 2009), survey trends from deep strata (24, 27, 28, 37-38, 29, 30, 36) were evaluated and found to be similar to the entire strata set (Figure C19). Diurnal/notcturnal comparisons showed no substantial differences between selected daytime and nighttime tows (Figure C20). No relationships were detected between survey catches and temperature (Figure C21).

The ASMFC-NEFSC summer shrimp survey samples shrimp habitat in the western Gulf of Maine (Figure C22). Data are available from this survey since 1985, and there have been no changes in vessel or gear. The summer shrimp survey catches pollock in a slightly greater proportion of tows than the NEFSC fall or spring surveys. Pollock lengths are measured on the summer survey, but age structures are not collected. The biomass trend from the summer survey is generally consistent with the fall survey in that biomass generally increased from the mid 1990s to 2004, but declined in recent years (Figure C23).

Pollock are also sampled by state surveys of inshore waters. The Maine-New Hampshire survey, in operation since about 2000, catches small pollock along the coast of Maine and New Hampshire in spring and fall. The Massachusetts survey, in operation since 1978, occasionally catches small pollock in spring, but few pollock are caught in the Massachusetts fall survey. State surveys may provide recruitment indices for the pollock assessment.

Relative abundance of pollock larvae from ichtyoplankton surveys may be considered as a proxy annual index of spawning stock biomass. An annual index of pollock larval abundance was derived using methods similar to those applied to herring by Richardson et al. (2010). Data from several sequential surveys were combined: 1971-1978 ICNAF, 1977-1988 MARMAP, 1989-1994 herring-sandlance survey, 1995-1999 GLOBEC, and 1999-2009 ECOMON. Each survey used a 61cm bongo net to sample to 200m deep, and up to 50 larvae were measured from each program. Mesh size was decreased from 505um to 330um in the GLOBEC survey. Pollock larvae were found from November to April, but primarily from December to March. The larval index suggests large spawning biomass in the mid 1980s, but much lower biomass since then (Figure C24). The WG noted the large difference in magnitude of the confidence intervals between the early and late period of the larval index time series. The difference in confidence intervals most likely results from different survey timing relative to the spawning season. The larval index was included in exploratory stock assessment models as an index of spawning biomass.

The WG decided that the MADMF inshore fall survey would not be considered as an index of abundance, because it catches too few pollock (e.g., pollock are not caught at all in many years). All other surveys (NEFSC spring, fall, summer and larval surveys; ME-NH inshore survey; MA spring inshore survey) would be evaluated as stock size indices in exploratory assessment analyses.

Age Structure –

Size and age structure from NEFSC spring and fall surveys suggest a relatively robust distribution of sizes and ages in the early 1970s, a truncation of large and old fish from the late 1970s to the turn of the century, with some rebuilding of size and age structure in the last decade (Figure C25, Tables C6a and C6b). With the exception of a relatively strong yearclass in the early 1970s, there is little correspondence among age-based survey indices to track yearclasses over time.

Stock Assessment

Term of Reference 3: Stock biomass, fishing mortality and recruitment

Estimate annual fishing mortality, recruitment and stock biomass (both total and spawning stock) for the time series, and characterize the uncertainty of those estimates.

Natural Mortality Assumption

Age data for pollock has been available since the early 1970s. The maximum age that has been seen in the NEFSC surveys since 1970 is 24 (Figure C26). There is no reason to believe that age structure was truncated before the mid-1970s, because removals during the 1970s and mid-1980s were three times the levels seen prior to 1970. The oldest age in the commercial age data is also 24, from a sample in 1984. An instantaneous annual natural mortality rate of 0.2 was used in previous assessments, and corresponds to approximately 1% survival to age 24.

Due to the lack of reliable data on natural mortality rate by age or year, it would be difficult to develop a time or age-varying mortality schedule. Although an age-specific mortality schedule could be developed using a functional response, the lack of data available to build such a model would make any gains from age-dependent mortality schedule negligible. The Working Group decided to assume M=0.2, because it is consistent with available data, and it was the value assumed in past assessments. The WG agreed that a sensitivity model run would consider M=0.15.

Size and Weight at Age

Data from surveys indicate that median age and mean length generally declined. Mean size at age plots showed some inter-annual variation for ages 1 to 10 (Figure C27a), with a slight decline suggested in recent years. Data for older fish are limited, and size at age estimates are more variable.

The WG decided that growth will be based on observed weight at age, and spawning weights will be based on January-1 weights using Rivard's interpolation method applied to the commercial catch weights. Weights at age show a consistent decline over the last decade (Figure C27b). Projections and reference points will be based on recent averages of weight at age.

Maturity

The 'hit or miss' nature of the pollock catches in surveys results in highly variable estimates of maturity at age resulting from low sample sizes in many years (Figure C28). When maturity data is pooled over all years, age 3 appears to be an inflection point in the maturity ogive, with most fish younger than 3 immature and most fish older than 3 mature (Figure C29). A time-averaged maturity leads to more reliable estimates of maturity at age. The WG decided that maturity at age will be assumed to be constant over time, and will be estimated using pooled-year data.

Update of Previous Assessment Method

Recent assessments of pollock applied an index-based method for "the portion of the unit stock of pollock primarily within the USA EEZ (NAFO Subareas 5 and 6) including a portion of eastern Georges Bank (Subdivision 5Zc) that is under Canadian management jurisdiction" (NEFSC 2002b). Overfishing was defined as the relative exploitation rate that allowed replacement, and B_{MSY} was approximated as the NEFSC fall survey biomass index from the 1980s (NEFSC 2002b). In 2006-2007, the fall survey index decreased (Figure C17), and the 2008 index-based assessment determined that the stock was overfished and overfishing was occurring (NEFSC 2008).

The most recent assessment used a centered three-year average for stock status determinations (NEFSC 2008). In order to provide catch advice for 2010 and 2011, the index-based assessment was updated with 2008 catch and survey data by the Multispecies Plan Development Team. The 2008 catch and 2007-2008 survey indices were used to 'project' the survey index value for 2009, however, this implied a negative survey index in 2009. As an alternative, the lowest observed fall survey index value was used to replace the implied negative 2009 value, and the 2007, 2008, estimated 2009 survey values were used to estimate the 2008 biomass proxy. While the pollock index from the fall survey is highly variable (even the log retransformed indices), projection results imply erratic fall survey indices and a pattern of a large increase in one year followed by two years of decline. When the lowest observed survey value is used for 2009, a two-year projection implies the survey value for 2010 will be near 0 and will increase by a factor of 37 in 2011. One reason that the projection gives unrealistic results is that it does not incorporate any stock dynamics—the method assumes that the stock will grow without interruption. The New England Scientific and Statistical Committee rejected the index-based assessment as a basis for catch advice in 2009.

To build a bridge between previous (AIM) and current (ASAP) assessment approaches, the AIM model was run with commercial landings through 2009 and the fall log-transformed index through 2009. The previous index biomass reference point (GARM III) was 2 kg/tow from the NEFSC Fall Bottom Trawl survey, and the previous overfishing reference point was 5.66. Using the data through 2009 for both landings and surveys the overfishing reference point estimate drops slightly to 5.41. The predicted MSY for the updated AIM assessment is 10,820 mt(ie.5.41(000mt/kg/tow)x2.0kg/tow).

The AIM model calculations of stock status and relative F were based on a 3-year centered average, so the most recent estimate with 3 observations corresponds to year 2008 (i.e., 2007-2009). The average survey abundance is 0.63 kg/tow. As this is lower than the previous biomass reference point of 2.0 kg/tow, the stock would be considered overfished. The average of the 2008 and 2009 survey estimates is 0.57 kg/tow and would also be considered overfished. The AIM model's relative replacement ratio estimate in 2008 of 0.6 indicates that the stock is declining at current values of relative F. The relative F estimated for 2008 is 16.3, which is about 3 times greater than the previous overfishing reference point 5.41. Theoretically the reference point relative F would keep the population at its current biomass. Therefore the AIM analyses would have concluded that overfishing was occurring.

There are numerous reasons why the two models (AIM and ASAP) reach different conclusions about stock status. First, the ASAP model includes age structure. This means that maturity, fecundity, and selectivity at age are incorporated in the ASAP framework. This is significant, because fishery selectivity has evolved from primarily selecting young immature fish to now selecting primarily large, mature fish. Additionally, while the fall index generally appeared to respond to trends induced by fishing, the last 10-15 years has seen a widening disparity between the selectivity of the fall index, which samples proportionately younger fish, and the fishery. The incorporation of the spring index, and the annual variances for both indices, allowed the model to properly smooth through trend without being driven by apparently large year effects. Finally, the ASAP assessment model takes a more complete accounting of total catch by including commercial discards, and recreational landings and discards.

Revised Assessment Method

Model Description

Pollock has been assessed using AIM (An Index Method, NEFSC 2002b) since 2000. Given the wide changes that have occurred in the fishery (gear, selectivity, targeting, and management), the change to a new survey vessel (for which a calibration cannot be estimated), the importance of age structure (maturity and growth), and the limited projection capability of AIM, alternative assessment methods were considered for this benchmark. The new assessment model is ASAP (Age Structured Assessment Program v2.0.20, Legault and Restrepo 1998), which can be obtained from the NOAA Fisheries Toolbox (http://nft.nefsc.noaa.gov/). As described at the NFT software website, ASAP is an age-structured model that uses forward computations assuming separability of fishing mortality into year and age components to estimate population sizes given observed catches, catch-at-age, and indices of abundance. Discards can be treated explicitly. The separability assumption is partially relaxed by allowing for fleet-specific computations and by allowing the selectivity at age to change in blocks of years. Weights are input for different components of the objective function which allows for configurations ranging from relatively simple age-structured production models to fully parameterized statistical catch at age models.

The objective function is the sum of the negative log-likelihood of the fit to various model components. Catch at age and survey age composition are modeled assuming a multinomial distribution, while most other model components are assumed to have lognormal error. Specifically, lognormal error is assumed for: total catch in weight by fleet, survey indices, stock recruit relationship, and annual deviations in fishing mortality. Recruitment deviations are also assumed to follow a lognormal distribution, with annual deviations estimated as a bounded vector to force them to sum to zero (this centers the predictions on the expected stock recruit relationship). For more technical details, the reader is referred to the technical manual (Legault 2008).

Model Inputs

Catch at age for years 1970-2009 are used for two distinct fleets: a composite commercial fleet, and a recreational fleet (Table C7a and C7b). The commercial fleet includes US catch by otter trawl and gillnet (with minor contributions from hook and line gear), as well as landings by distant water fleets (1970-1976) and Canadian fleets (1970-1985). Total discards for the commercial fleet are estimated for years 1989-2008 from observer data. Discards at age were estimated from discard length frequencies, raised by estimated total discards by area and gear (otter trawl, gillnet). Age length keys from combined survey and commercial data were used to obtain number at age from number at length. Data were not available to estimate discards for 2009, so it was assumed that total mt of discards in 2009 were the same as in 2008, and no age composition was included in the objective function for 2009.

Catch for the recreational fleet begins in 1981 when a standard method of data collection and statistical estimation was initiated (*Marine Recreational Fisheries Statistics Survey, MRFSS*). Landings and discards are assumed to have the same length frequency, and discard mortality is assumed to be 100%. Expanded length frequencies were converted to catch at age by multiplying by age length keys from survey data.

Several model runs were performed with a sensitivity assessment model (SCAA by Butterworth and Rademeyer, see below) including one or more of the sensitivity indices (NEFSC summer, NEFSC larval, ME-NH spring and fall, MA spring). Examination of these runs suggested that the sensitivity indices were not adding information or signal to the model estimated trends. Furthermore, the WG felt that the assumed selectivities for these indices, which required an assumption about size at age by season for young fish, needed a more detailed analysis due to the rapid growth realized by fish aged 1 to 3. The WG decided that these indices should be considered in future assessments if the lengths could be treated suitably. Consequently, only the NEFSC Spring and Fall surveys were used in the model. Annual number/tow and the estimated CV were used along with annual estimated age composition for years 1970-2009.

Age-specific but time invariant maturity was used in the model. An age and time invariant natural mortality (M) of 0.2 was assumed.

Base Model Configuration (ASAP)

Model estimates of selectivity at age were freely estimated for fisheries and surveys, with no restriction for flat-topped or dome-shaped results. Although it is difficult to directly observe relative selectivity of old ages, domed selectivity for pollock can be justified from information on fishing gears and pollock behavior. Gillnets, which contribute approximately 40% of the recent commercial landings, typically have dome-shaped selectivity (Hamley 1975), and gillnet selectivity of pollock was estimated to be dome shaped in the Gulf of Maine (Marciano et al. 2005). Pollock also have greater swimming speed and endurance than other groundfish, and swimming speed increases as a function of size (He and Wardle 1988). Therefore, selectivities that have a dome-shape (i.e., selectivity at older ages is <100%) would not be an unexpected result. Furthermore, it is worth noting that the selectivity estimated for the 9+ group reflects the catchability for all ages 9 and older.

Beginning with a single selectivity function for each fleet, model diagnostics were examined for trends in age composition residuals. With only one selectivity vector per fleet, there were strong trends in residuals with long runs of positives and negatives (Figure C30).

Additional selectivity blocks were added one at a time, with each fleet being addressed separately, until residual patterns were acceptable. The addition of selectivity blocks was balanced against the reduction in the objective function value (given the added parameters) to avoid overparameterization. To determine the best year for introducing new selectivity blocks, a split was introduced for several consecutive years and the model with the lowest objective function value determined the year when the new block would begin. Somewhat concurrent with this process, changes in fleet composition (e.g., following the establishment of the EEZ in 1976, establishment of The Hague Line in 1985) and major management changes (such as introduction of minimum sizes, changes in mesh size and introduction of closed areas), were considered as potential years where a new selectivity block might be anticipated.

The base model contains four selectivity blocks for the commercial fleet with breaks between the following years: 1985/1986, 1993/1994, 2003/2004. The 1985/1986 split can be related to the international boundary decision, with recent commercial catch at age coming exclusively from the US fleets rather than including foreign fleets. Furthermore, a 17 inch minimum size was introduced (previously there had been no minimum size), and a minimum mesh size of 5 ½ inches was introduced for sink gillnet fishing in the mid 1980s. The 1993/1994 block can be related to an increase in trawl mesh size from 5 ½ to 6 inches, and the year round closure of Closed Areas I and II. There were numerous management actions between 2001-2004, including increasing trawl mesh and sink gillnet mesh sizes to 6 ½ inches, and differential days at sea counting. Each consecutive selectivity vector shows a trend towards selecting older fish, which appears to be consistent with management regulations (Figure C31).

For the commercial fleet, selectivity at age is estimated within each block for 8 out of 9 ages, with one age class fixed at full selectivity in each block. In the interval 1970-1985, selectivity at age 6 is assumed fully selected, while in the remaining blocks age 7 is assumed fully selected. The estimated selectivities are dome shaped, and while a double-logistic form would have been more parsimonious, freely estimating selectivity at age was chosen over estimating selectivity with a double logistic due to convergence problems. Estimates for the parameter defining the age of 50% selectivity for the descending limb were tending towards the plus group (age 9), leading to boundary solutions or simply lack of convergence. Expanding the catch at age so that the plus group occurred at age 12 resolved the boundary problem (unless the descending a50 was fixed at 12), but the working group felt that the data at that age were too sparse and the model would more likely be fitting noise rather than signal.

Three selectivity blocks are estimated for the recreational fleet with breaks occurring between the following years: 1993/1994, 2001/2002. Selectivity in each period was estimated with a double logistic function and there were no problems with parameters being estimated at boundaries. No specific management or fleet change occurred in 1993-1994, although a federal minimum size of 19 inches was introduced for recreational fishing in 1989. As fish continued to be landed below the federal minimum size, this regulation is not believed to have had a significant effect on landing patterns, partly from the lack of minimum size regulations in state waters. The selectivity block in 2001/2002 reflects a shift in the mode of fishing that accounted for the greatest proportion of catch. Previously, the shore mode had contributed on average about 20% of the catch, although in any given year it ranged from 5% to 65%. After 2001, the shore mode of fishing contributed 5% or less, while the rest of the catch was contributed by private/rental boats or by party/charter boats. As the shore mode includes fishing from the beach, piers, bridges, and other fixed structures, this mode primarily catches what are referred to as ‘harbor pollock’—principally fish aged 1-3 (Figure C32). The selectivity estimated for the final block is shifted towards older ages, which seems consistent with the change in mode of fishing, and may reflect greater adherence to the federal minimum size.

One time invariant selectivity vector was estimated for each of the two surveys (NEFSC Spring and Fall). Selectivity was estimated freely for 6 out of 9 ages for both the spring and the fall survey, with the remaining three ages fixed: ages 6 and 7 were assumed to be fully selected, and age 9 was fixed at a value of 0.5 (Figure C33). When selectivity at age 9+ was freely estimated, the model estimated a value of 0.25 for the spring and 0.22 for the fall index. However, such a sharp dome implied that starting spawning stock biomass in 1970 was nearly 3 times greater than the deterministic estimate of unexploited spawning biomass, which was not believed to be realistic. A fixed value of 0.5 was accepted by the working group after trying values from 0.1 to 1.0 (in increments of 0.1) and examining model diagnostics (residual patterns in age composition for both surveys and catch), objective function value, and the reasonableness of estimated abundance levels. The abundance levels were evaluated by examining the model estimate of the ratio of initial spawning biomass to unexploited spawning biomass (SSB1970/SSB0), and inspecting the time series of estimated SSB relative to a heuristic 'envelope' of realistic biomass levels (described more fully below). The model estimate of steepness was another diagnostic, and runs that estimated steepness near its upper bound of 1.0 were dropped from further consideration. This series of diagnostics reduced the set of values considered for selectivity at ages 9+ to 0.3, 0.5, and 0.6, although the initial spawning biomass with 9+ selectivity of 0.3 was somewhat high at double the unexploited SSB. Retrospective analysis for the 7 preceding years (2002-2008) was then performed for models using each selectivity value. The model with index selectivity fixed at 0.5 or 0.6 achieved convergence for 6 out of 7 runs, with logical retrospective patterns (Figure C34). Only 5 out of 7 runs with selectivity fixed at 0.3 converged. Needing to proceed with an approach which readily provided convergence across other retrospective runs, the working group adopted the model with selectivity fixed at 0.5 as the base formulation.

The effective sample size estimated for the catch at age data (which are treated as multinomial) was compared to the input effective sample size in an iterative fashion until the effective sample size specified more or less matched the model estimated value, or until no further improvement in trying to match the estimated value could be made. The final input effective sample sizes were 50 and 35 for the commercial and recreational fleets, respectively. An annual CV of 0.05 and 0.25 were assumed for the commercial and recreational landings, respectively. Commercial discard CVs for 1989 to 2008 were estimated as part of the standardized bycatch methodology. These values ranged from 0.12 to 1.04, with an average of 0.33. The estimated annual CV for recreational discards ranged from 0.47 to 0.91, with an average of 0.67.

In a similar fashion, the input effective sample size for the survey catch at age was manually tuned until the model estimate was reasonably close to the input value. For both surveys, the final input effective sample size was 30. The annual CV for each survey was the design based estimate (the surveys follow a stratified random design). For the spring survey, the average CV for the time series is 0.37, although it ranges from 0.18 to 0.85. For the fall survey, the average CV for the time series is 0.42, with a range of 0.19 to 0.74. These CVs reflect the strong year effects present in the survey.

Recruitment was assumed to follow a Beverton-Holt functional form, with an assumed CV=0.5 for annual recruitment deviations (i.e. on log-space the standard deviation of the residuals about the stock-recruitment relationship was 0.5).

Spawning was assumed to occur January 1. This is consistent with observations that the peak spawning period occurs December-January. Initially, observed lengths at age in the spring survey were used to calculate spring weight at age, and spring weights were used to estimate January 1 weights at age by the Rivard method. However, there was considerable variability between and within cohorts, and in many cases cohorts appeared to lose weight with age. The working group decided to use the observed catch weights at age, treat them as mid-year weights, and use the Rivard method to obtain January 1 weights at age. These new 'Rivard-ed' catch weights were then used as the spawning weights at age.

Base Model Results

Biomass –

The base model estimates a starting spawning stock biomass (SSB) in 1970 of about 297,000 mt, which is approximately 9% above the deterministic, point estimate of unexploited spawning biomass (~273,000 mt). Spawning biomass decreased to the time series low (68,600 mt) in 1990 (Table C8, Figure C35). Since the 1990 low, spawning biomass increased steadily through 2006, with a slight decline the last 3 years. The current estimate of spawning biomass is about 196,000 mt.

Two additional biomass measures were calculated from the estimated numbers at age (Table C9). Total population biomass was calculated with January 1 weights at age while exploitable biomass was calculated with mid-year catch weights at age and annual selectivity at age (Tables C10a,b). Total population biomass follows the same trend as SSB (Table C11, Figure C35). Exploitable biomass ranges from 35% to 70% of spawning biomass over the time series (Table C12). Due to the estimated dome-shaped fishery selectivities, exploitable biomass will always be less than spawning biomass.

Fishing Mortality –

In any given year, the fishing mortality experienced by an age class depends on the selectivity and amount of catch of each fleet. To provide a consistent metric for expressing F over the whole time series, the unweighted average F for ages 5-7 (F_{5-7}) is reported (Table C13). In 1970, F_{5-7} is estimated at 0.11, and mostly increased to its peak of 0.49 in 1986. Since then, F_{5-7} steadily decreased to 2006, when it reached the time series low of 0.03. In the last three years, F_{5-7} was 0.05, 0.08, and 0.07, respectively.

Recruitment –

Mean recruitment was around 21 million age 1 recruits. Several abundant year classes were produced in 1971, 1979, 1997, 1998, 1999, and 2001, with the estimated number at age ranging from 34 to 58 million (Figure C36). The model estimated steepness at 0.66 with a CV of 0.24 (Figure C37).

Catch –

As a result of the small CVs assigned to the commercial landings, they were well fit (Figure C38). Commercial discards, which used CVs estimated from the data, had larger residuals compared to the landings (Figure C39). Increasing the number of selectivity blocks from one to four vastly improved the residuals in the commercial age composition (Figure C40). The final input effective sample size approximately matches most of the model estimated effective sample sizes (Figure C41).

The CV assigned to the recreational landings was five times greater than the commercial landings CV (0.25 versus 0.05), but they were still fit well (Figure C42). Recreational discards, which used CVs derived from the recreational landings data, had larger residuals compared to the landings (Figure C43). Increasing the number of selectivity blocks from one to three improved the residuals in the recreational age composition (Figure C44). The final input effective sample size does a reasonable job of matching most of the model estimated effective sample sizes (Figure C45).

Indices –

As noted above, the indices show apparently strong year effects, but these years tended to have the largest CVs. Thus, in fitting the indices, the influence of these effects was not strong. The predicted spring index smoothes through the early and late part of the time series, but there is a stretch of positive residuals in the 1980s and 1990s (Figure C46). The residuals in the spring age composition show some persistent trends at age for several year blocks, although the year-age blocks with the trends do not appear to be related (Figure C47). The age composition of the indices was downweighted relative to the landings by having a lower effective sample size (30, versus 50 and 35 for the commercial and recreational fleets, respectively). Although Figure C48 suggests that the indices could be downweighted further, this was not pursued.

The predicted fall index smoothes through the time series until about 1990, when there is a run of positive residuals through 2006 (Figure C49). The residuals in the fall age composition show some persistent trends at age for several year blocks (Figure C50). Unlike for the spring, however, these residual blocks somewhat trace diagonals through the plot and may reflect cohort effects. As was the case for the spring index, Figure C51 suggests that the fall index could be downweighted further but not to the extent that was seen for the spring index. Further downweighting was not pursued.

Envelope Analysis

An ‘envelope analysis’ was presented at the model meeting as a simple method to bound reasonable abundance estimates. The time series of total catch (mt), spring index (kg/tow), and fall index (kg/tow) were converted to total population biomass as follows:

$$Biomass(Catch) = Catch(y) / F$$

$$Biomass(SpringIndex) = SpringIndex(y) \times q_{Spring} \times A_{swept} / A_{tow} / 1000 .$$

$$Biomass(FallIndex) = FallIndex(y) \times q_{Fall} \times A_{swept} / A_{tow} / 1000$$

In the above, A_{swept} is the total area in the survey stratum (33,192 nm) and A_{tow} is the area swept by a tow (0.01 nm); these are divided by 1000 to maintain biomass units in mt. Index specific catchabilities are denoted q_{Spring} and q_{Fall}. Note that these equations tacitly assume full selectivity at all ages in the catch and the surveys.

For each biomass time series, a low and a high bound was calculated by assuming 2 values for F or q. In this particular analysis, the values considered were F={0.05, 1.0}, q={0.05, 0.50}. While these values weren’t necessarily data-driven, assuming an F of 0.05 for all years would likely overestimate maximum abundance in some years and underestimate maximum abundance in other years. Similarly, assuming a q of 0.05 assumes fairly low catchability for the surveys. If catchability were actually lower, then the biomass calculated from q=0.05 would underestimate the maximum annual abundance. With these caveats in mind, the minimum and maximum biomass over the set of 3 biomass time series were plotted for each year to suggest reasonable bounds against which model estimated biomass could be compared. Figure C52 shows the envelope with 3 different biomass measures calculated from the new base model:

Total Biomass = $\sum_{age=1}^{9+} N_{age} W_{age,Jan1}$

Spawning Stock Biomass = $\sum_{age=1}^{9+} N_{age} W_{age,Jan1} p_{age}$

Exploitable Biomass = $\sum_{age=1}^{9+} N_{age} W_{age,Mid-yr} sel_{age}$

In the above, p_{age} is the proportion mature at age, and sel_{age} is the age-specific selectivity across both fleets. Note that both total biomass and spawning stock biomass used January 1 weight at age, while the exploitable biomass used mid-year weight at age.
This heuristic exercise provides further support that the ASAP base model abundance estimates are not unreasonable.

Retrospective analysis

Retrospective analysis was performed for years 2002-2007 (7 years). Before all selectivity blocks had been added to the model, the working group discussed whether retrospective analyses should be considered if selectivity changed in the most recent 7 years. The base model has recreational selectivity changing between 2001/2002, and the commercial fleet selectivity changes between 2003/2004. The working group suspected that changing selectivity during the years analyzed for retrospective analysis might tend to inflate the pattern as the model attempted to estimate selectivity parameters with fewer and fewer years of data. The pattern in Figure C34 shows two distinct clusters in the retrospective pattern for F_{5-7} and SSB. The earliest years, which encompasses the change in recreational selectivity (2002-2003), is clustered furthest away from the origin (i.e., those years have higher relative retrospective bias). The years following the change in commercial selectivity are clustered (2004-2005), while the most recent three years (2006-2008) are much closer to the origin (lower relative retrospective bias). The working group interpreted this pattern as the model needing enough years beyond the last selectivity changes in order to reliably estimate those selectivity parameters. If all seven years are used to calculate Mohn's rho (the 7 year average of relative retrospective bias), then the values are -0.17 for F_{5-7} and 0.27 for SSB; using only 2006-2008 retrospective values, the average bias is -0.08 for F_{5-7} and 0.13 for SSB. The average retrospective bias for 2006-2008 is small relative to other groundfish assessments in the Northeast.

MCMC simulation

MCMC simulation was performed to obtain posterior distributions of spawning stock biomass and F_{5-7} time series. Two options in ADMB were invoked to reduce high autocorrelation. The variance-covariance was rescaled (with mcrb 2), and the tails of the sampled distribution were "fattened" (with mcgrope 0.07) (ADMB 2008). Initial trials without rescaling or without fattening the tails produced traces that resembled random walks rather than random sampling, i.e. there was high autocorrelation and strong evidence that the chains were

not well mixed. Two chains of initial length 10 million were simulated. The first half of each chain was dropped, and from the second half of the chain every 5,000th value saved, producing two chains of length 1,000. The traces of each chain's saved draws were plotted, and both indicated good mixing (Figure C53). Autocorrelations for F_{5-7} ranged from 0.26 in 1970 to 0.37 in 2009 with a lag of 1, and were less than 0.22 with a lag of 2 or greater. Autocorrelation for SSB ranged from 0.27 to 0.54 with a lag of 1, and were <0.4 with a lag of 2, <0.3 with a lag of 3, and < 0.24 with a lag of 4. The decreasing autocorrelation with increasing lag is another good indicator that the MCMC chains have converged. Finally, the Gelman-Rubin potential scale reduction factor (psrf) was calculated for the time series of F_{5-7} and SSB. All psrf were between 1.0 and 1.01, which again suggests convergence of the chains.
As the MCMC simulations appear to have converged, 90% Probability Intervals were calculated to provide a measure of uncertainty for the model point estimates (Figures C54, C55). Plots of the posterior for SSB_{1970}, SSB_{2009} and $F_{5-7(2009)}$ are shown for both chains in order to characterize the density of each distribution (Figures C56a-b, C57).

Sensitivity analysis of ASAP base model

A sensitivity model was examined where selectivity in both the spring and fall NEFSC surveys was fixed at 1.0 for ages 6-9+. The effect of this was predictable, in that abundances were scaled lower. Specifically, SSB in 1970 was 94,000 mt instead of 297,000 mt. Also, current biomass with flat survey selectivity dropped to 77,000 mt from 196,000 mt in the base model. Model estimates and likelihood components are compared in Table C14 for the ASAP base model, for this sensitivity model with index selectivity fixed at 1.0 for ages 6-9, and for the converged models where the index selectivity for the 9+ group was varied between 0.1-1.0. Compared to the base model, the age composition residuals for both the indices and the fleets barely changed. However, the fits to the indices were worse, with the indices dropping even further below the observed values from the 1990s and later. A retrospective run of the model with flat survey selectivities led to one year where the model couldn't run to completion (2003). For the remaining 6 years, the retrospective pattern had relative biases that were more than twice as poor as the base case (Figure C58). The 6 year average Mohn's rho for F was -0.41, and the 3 year average was -0.26. For SSB, the 6 year average Mohn's rho was 1.06, and the 3 year average was 0.54.

A sensitivity model was examined where natural mortality (M) was fixed at 0.15 instead of 0.2 for all ages and all years. The result of a lower M was to increase the estimated depletion through time, such that in 2009, spawning biomass was 45% of unexploited SSB instead of 72% under the base model. Lowering M to 0.15 increased the objective function value by 9 points over the base model.

As a simple exploration of the impact of using only the catch in US waters of NAFO areas 5 and 6, Canadian landings on the northeast corner of Georges Bank (5Zc, Figure C1) were included in the time series of total commercial landings (Table C15). No landings were reported by Canada in this area before 1982. The fraction of landings by Canada in 5Zc were generally less than 20% of total commercial landings with the exception of a period from 1992-2005, when Canadian landings ranged from 22% to 47% of the total. In the most recent 3 years, Canadian landings in 5Zc have been minor. It was assumed that these landings would have the same size/age structure, so catch at age was simply scaled to reflect the increase in total landings. No discarding was assumed for Canada in 5Zc. The effect on model results was minor. Estimates of initial conditions in 1970 were generally 4% less than the base model, while estimates for 2009 were 9% less (Table C14).

Sensitivity analysis to assessment model (Butterworth & Rademeyer SCAA)

An additional statistical catch at age (SCAA) assessment model was considered during the working group model meeting (29 March – 2 April, 2010). This model, the mathematical details of which are given in Appendix C2, differs from ASAP in several ways.

- The initial numbers-at-age vector was not estimated for all ages, but instead represented more parsimoniously in terms of two estimable parameters: • – the starting spawning biomass as a proportion of the corresponding deterministic pre-exploitation level, and ϕ reflecting an average fishing mortality (see equations B8 to B12 in Appendix C2). In implementation, the starting year chosen was 1960 rather than the 1970 for ASAP, so that a few more years of the early survey data were fitted. Furthermore the priors for • and ϕ for computing posterior distributions by means of MCMC were chosen as U[0.2;1.2] and U[0;0.3] respectively.
- Pope’s approximation rather than the Baranov equation was used for the dynamics to speed computations, though the consequent differences would be rather small.
- In fitting to the survey indices of abundance, the inverse variance weighting approach used in computing the likelihood took account of an estimable additional variance as well as the sampling variance estimates that accompanied the survey data (see equations B18 and B19 with associated text in Appendix C2).
- Rather than a multinomial distributional form assumed for commercial or survey proportions-at-age data when computing the likelihood in ASAP, a modified log-normal was used with the intent of capturing both process and sampling error effects in a parsimonious way (see equations B20 to B24 in Appendix C2). The associated variance parameter was estimated directly from the residuals in the fitting procedure. Customarily such contributions to the negative log-likelihood are downweighted to allow for non-independence amongst such data inputs; here a multiplicative downweighting factor (w_{CAA}) of 0.1 was used, though runs without this downweighting were also conducted.
- A greater differentiation among fleets was effected with six distinct “fleets” being distinguished: US, distant water, and Canadian commercial fleets, as well as commercial discards, recreational landings and recreational discards.
- The selectivity functions (from models with a plus group at age 9+) were differently specified compared to ASAP. Selectivities were invariant over time unless selectivity “blocks” (see below) were specified for a particular “fleet”. For each (block for each) “fleet”, selectivity was estimated directly for each age from age '*data-minus*' to age '*data-plus*', where data were grouped below and above such ages when fitting to the model because of sample size considerations. The estimated decreases from ages *data-minus*+1 to *data-minus* and from ages *data-plu*s-1 to *data-plus* were assumed to continue exponentially to ages 1 and 9 (the model plus group considered) respectively. For the commercial fisheries *data-minus* was taken to be 3, and 1 for the other “fleets”, while *data-plus* was set at 9 for the US commercial, 8 for the other commercial and the recreational, and 6 for both discard “fleets”. For the NEFSC spring and fall surveys, the fishing selectivity was estimated directly for each age from age 1 to age 8 and to age 7 for

the spring and fall surveys respectively, and was assumed to remain constant at those age 8 and age 7 values for higher ages.

During the model meeting, extensive testing of both models occurred. At the close of the model meeting, the working group felt comfortable that despite the structural differences between the two models, they were capable of producing similar results when configured similarly. Thus, the SCAA model provided valuable feedback regarding model sensitivity to assumed error distributions, estimation of starting conditions, and selectivity fitting.

As not all model inputs were complete by the model meeting, subsequent runs of this *SCAA* were conducted with the full data set (the same as used in the ASAP base model, as described above). To the extent possible, the SCAA was configured to match the ASAP base model to cross-check results. There were nevertheless some differences because of time limitations, though indications are that the impact of those differences on results would be small:

- The choice of periods (blocking) during which selectivity for a “fleet” remained the same differed from the ASAP implementation by including one extra selectivity block for the US commercial fleet, with the first block used for the ASAP model being split in 1976/1977. For the recreational “fleet”, the first block was split in 1989/1990 instead of 1993/1994 as for ASAP.
- The Beverton-Holt stock-recruitment function steepness estimate was bounded above by 0.9.
- All catches (commercial, discard and recreational) were fixed on input without allowing the model fitting process to select possible relatively small errors in each year.

Table C16 compares results for some key outputs from the SCAA approach to those from the base case ASAP run. The SCAA runs shown converged reasonably, both in respect of point estimate and Bayes posterior computations achieved using MCMC. The runs commenced in 1960, and did not typically reflect values of SSB in 1970 greater than SSB0. Results are shown for three SCAA implementations, with the specifications detailed above, and compared with those for the ASAP base case in Table 16:

- SCAA1 downweights the CAA data ($w_{CAA} = 0.1$).
- SCAA2 gives full weight to the CAA data ($w_{CAA} = 1$).
- SCAA3 duplicates SCAA2, except that in the MCMC the selectivity of 9+ fish in the surveys is fixed at the point estimate for SCAA2.

SCAA2 is likely the closer analog of the ASAP base case in terms of the relative weight given to CAA data in the model fitting process, and associated point of MCMC estimates for SSB for this run are shown in Figure C59. SCAA3 is closer to the ASAP base case prescription in terms of variance computation, as it fixes the 9+ survey selectivity as in the ASAP case.

The SSBMSY and MSY estimates shown in Table C16 are not evaluated using the Beverton-Holt stock-recruitment curves estimated in these model fits, but instead are proxies based on $F_{40\%}$. They differ slightly in methodological terms from corresponding values calculated for the ASAP runs in that they reflect the multiplication of estimates of SSB/R and Y/R at $F_{40\%}$ by the average recruitment (which here is as estimated for the 1970-2005 period). Any changes in estimates of these proxies as a result of this difference should however be small.

In broad terms, these SCAA runs show very similar historic trends in spawning biomass to those from the base case ASAP. Both the scale (average magnitude over time) and the variance associated with the spawning biomass estimates are however larger for the SCAA runs than for the base case ASAP. Much of this difference relates to the weighting given to the CAA data in the model fit. As this weight is increased, both posterior medians and 95%-iles decrease to become closer to the ASAP estimates. However, even if the 9+ survey selectivity is fixed at its value in SCAA2 when estimating variance, results for spawning biomass still reflect less precision than do those for the ASAP base case. Nonetheless this scale difference translates only slightly (if at all) into estimates of sustainable yield, with MSY proxy estimates and their precision for SCAA2 and SCAA3 broadly similar to the results obtained from the ASAP base case.

Management Reference Points

Term of Reference 4: Update or redefine biological reference points
(BRPs; estimates or proxies for B_{MSY}, $B_{THRESHOLD}$, and F_{MSY}; and estimates of their uncertainty). Comment on the scientific adequacy of existing and redefined BRPs.

The working group decided to adopt F40% as a proxy for F_{MSY}. The NOAA Toolbox program YPR was used to calculate a deterministic value for F40% given average vectors for the most recent 5 years (2005-2009) for SSB weights at age, catch weights at age, maturity at age (which is time invariant), and selectivity at age. Expressed as the average F experienced at ages 5-7, the estimate is $F40\%_{5\text{-}7}$ = 0.25, which corresponds to a fully selected F of 0.41.

The population numbers at age for year 2010 corresponding to each saved draw from one of the MCMC chains were used to make stochastic projections to determine the SSB and yield corresponding to F40%. In the stochastic projections, recruitment was resampled from the empirical distribution as estimated by the ASAP base model for years (1970-2007). The stochastic projections were made using the NOAA Toolbox program AGEPRO, and each projection was made for 100 years to allow the projection to reach equilibrium.

From the projected distributions of SSB and yield, the median value was taken as the proxy for SSB_{MSY} and MSY. The proxy for SSB_{MSY} is 91,000 metric tons, with 5th and 95th percentiles spanning 71,000 to 118,000 mt. One half of SSB_{MSY} is the $B_{THRESHOLD}$ (45,500 mt). The proxy for MSY is 16,200 mt, with 5th and 95th percentiles spanning 11,800 to 23,200 mt. It should be noted that the MSY estimate includes both commercial and recreational landings and discards. The median recruitment was 19.3 million age 1 fish, with 5th and 95th percentiles ranging from 8.4 to 42 million fish. Distributions for SSB_{MSY} and MSY are given in Figure C60.

A second stochastic projection was done for 0.75* $F40\%_{5\text{-}7}$ = 0.19, which corresponds to a fully selected F of 0.31. Spawning biomass under a harvest at 0.75* $F40\%_{5\text{-}7}$ has a median of 109,000 mt, with 5th and 95th percentiles ranging from 86,000 to 140,000 mt. The corresponding median yield is 14,500 mt, with 5th and 95th percentiles ranging from 10,700 mt to 20,600 mt. The distribution of recruitment is independent of the harvest scenario, as it is merely sampling from the cdf of estimated values from the base model. Thus, the median recruitment was still 19.2 million age 1 fish, with 5th and 95th percentiles ranging from 8.4 to 42 million fish.

To evaluate the sensitivity of reference points to the model estimated dome-shaped selectivities, results from the flat-topped sensitivity model run were also used to estimate reference points. Following the same methodology, the average F40% on ages 5 to 7 was 0.22, the proxy for SSB_{MSY} was 58,000 mt, and the proxy MSY was 11,200 mt. Thus, if the survey selectivity at ages 6-9 is fixed at 1.0, rather than having a dome shape, then the biomass reference points would be 30-35% lower.

Stock Status

Term of Reference 5: Evaluate stock status with respect to the existing BRPs. *as well as with respect to updated or redefined BRPs (from TOR 4).*

The estimate of $F_{5\text{-}7}$ in 2009 from the ASAP base model (0.07) is 28% of the F_{MSY} proxy for ages 5 to 7 (0.25). Therefore, overfishing is not occurring. To provide a historical perspective on overfishing, a time series of $F_{40\%}$ corresponding to a fully selected F is plotted in Figure C61. This year-specific $F_{40\%}$ was calculated for years 1974-2009 with a 5 year moving average of weights at age, selectivity at age, and maturity at age. The $F_{40\%}$ in 1974 used years (1970-1974) while the final $F_{40\%}$ used years (2005-2009). The reason for doing this is that selectivity at age has changed substantially through time (Figure C62), and an $F_{40\%}$ in recent years when fishing occurs on mature fish would not be an appropriate reference point earlier in the time series when fishing occurred on immature fish. The calculated $F_{40\%}$ on ages 5-7 ranges from a low of 0.20 in 1976 to a high of 0.28 for 2000-2003. Considering the year-specific $F_{40\%}$ estimates, the base model estimates of F indicates that overfishing was occurring during the period 1973-1990.

The estimate of SSB in 2009 from the ASAP base model (196 000 t) is more than twice the SSBmsy proxy (91 000 t). One half of SSB_{MSY} is the $B_{THRESHOLD}$ (45,500 mt). Therefore the stock is not overfished. Similar to the reasoning above for F40%, the SSB_{MSY} proxy calculated using recent selectivity and weight patterns is not appropriate to compare to historic estimates of SSB. The year-specific $F_{40\%}$ values were used to make stochastic projections for determining the median equilibrium SSB_{MSY}. The full time series of model estimated recruitments was used in all projections, even for the 1974 estimate of SSB_{MSY} when the model would theoretically have only had 5 years of observations. The estimated year specific SSB_{MSY} proxies range from 91,000 mt to 122,000 mt, and indicate that the base model estimates of SSB $<$ SSB_{MSY} during the period 1987-1998 (Figure C63).

This revised assessment provides a different perception of stock status when compared to the stock status results from the AIM model. The most recent update of the AIM model indicated that the stock was overfished and overfishing was occurring in 2008. As Figure C64 indicates, the divergence between the NEFSC fall index selectivity and the fishery selectivity is especially pronounced towards the end of the time series. This divergence is important, as the AIM model assumes that the selectivity is the same in the fishery and the index.

The sensitivity of stock status to the model estimated dome-shaped selectivities was evaluated by comparing current F and SSB estimates from the sensitivity model with flat survey selectivity for ages 6-9 to their corresponding reference points. Assuming flat survey selectivity, the model estimate of SSB_{2009} was 77,000 mt, which is greater than the SSB_{MSY} proxy of 58,000 mt, so the stock would not be considered overfished. The model estimate of $F_{5\text{-}7}$ in 2009, assuming flat survey selectivity, is 0.13, which is less than the corresponding F40% on ages 5-7 of 0.22, so overfishing is not occurring. It was therefore concluded that stock status is not sensitive to the shape of survey selectivity at older ages.

Projections

Term of Reference 7: Develop and apply analytical approaches and data *that can be used for conducting single and multi-year stock projections and for computing candidate ABCs (Acceptable Biological Catch).*

a) *Provide numerical short-term projections (through 2017). Each projection should estimate and report annual probabilities of exceeding threshold BRPs for F, and probabilities of falling below threshold BRPs for biomass. In carrying out projections, consider a range of assumptions to examine important sources of uncertainty in the assessment.*
b) *Comment on which projections seem most realistic, taking into consideration uncertainties in the assessment.*
c) *For a range of candidate ABC scenarios, compute probabilities of rebuilding the stock by 2017.*
d) *Describe this stock's vulnerability to becoming overfished, and how this could affect the choice of ABC.*

The base ASAP model estimates that the stock is not overfished, so no rebuilding projections were conducted. However, for the purposes of providing advice for setting ABCs, the projections described above ($F=F_{40\%}$, and $F=0.75*F_{40\%}$) are summarized through 2017. In addition, a third projection, $F_{status\text{-}quo}$ was conducted with the same bootstrapped numbers at age and the same recruitments, but F was fixed at $F_{2009}=0.12$ (equivalent to $F_{5\text{-}7}=0.07$).

Projections are summarized for various percentiles of spawning stock biomass and catch under all 3 scenarios in Tables C17a, b. Under all three scenarios, spawning biomass declines from SSB_{2009}=196,000 mt until it reaches equilibrium at the projected F. Under $F_{status\text{-}quo}$, the median SSB equilibrates at 166,000 mt. Projecting at $0.75*F_{40\%}$, the median SSB equilibrates at 109,000 mt, while at $F_{40\%}$ the median SSB equilibrates at 91,000 mt (the proxy for SSB_{MSY}).

Projected catch includes both commercial and recreational landings and discards. Under $F_{status\text{-}quo}$, median projected catch decreases from 8,100 mt in 2010 to 7,200 mt in 2012, then gradually increases until equilibrating around 8,400 mt in 2017 (Table C17b). Projecting at $0.75*F_{40\%}$, the median catch fluctuates from 19,800 mt in 2010 to 15,400 mt in 2012, and continues to oscillate in this range until equilibrating at 14,500 mt. Projecting at $F_{40\%}$, median catch declines from 25,700 mt in 2010 to 17,500 mt in 2017 with minor fluctuations until equilibrating at 16,200 mt (the proxy for MSY). It should be noted that a projected 2010 catch of 25,700 mt would exceed MSY, be more than double recent catch, and has not been observed since the 1980s.

Trophic Ecology

Term of Reference 6: Evaluate pollock diet composition data *and its implications for population level consumption by pollock.*

Food habits were evaluated for pollock as a major predator in the ecosystem. The total amount of food eaten and the type of food eaten were the primary food habits data examined. From these basic food habits data, diet composition, per capita consumption, total consumption, and the amount of prey removed by pollock were calculated. Contrasts to total energy flows in the ecosystem and fishery removals of commercially targeted skate prey were conducted to fully address the Term of Reference.

To estimate mean stomach contents (S_i), pollock had the total amount of food eaten (as observed from food habits sampling) calculated for each size class, temporal and/or spatial scheme. The denominator in the mean stomach contents (i.e., the number of stomachs sampled) was inclusive of empty stomachs. These means were weighted by the number of tows in a temporal and spatial scheme as part of a two-stage cluster design. Further particulars of these estimators can be found in Link and Almeida (2000). Units for this estimate are in g.

Estimates were calculated on an annual basis for each pollock size class. These size classes corresponded to < and • 50 cm for Small (S) and Large (L) size classes, respectively. The food habits data collections started quantitatively in 1973. For more details on the food habits sampling protocols and approaches, see Link and Almeida (2000). This sampling program was a part of the NEFSC bottom trawl survey program; for background and context, further details of the survey program can be found in Azarovitz (1981) and NEFC (1988). Key diagnostics were the number of empty stomachs over time and mean length vs. mean stomach contents weight (with ± 95% CI), which were examined to identify any major outliers in the data and to ascertain any notable patterns in variance.

To estimate diet composition (D_{ij}), the amount of each prey item was summed across all pollock stomachs. These estimates were then divided by the total amount of food eaten in a size class, temporal and spatial scheme, totaling 100%. These estimates are proportions and were only presented for those major prey comprising >85% of the total for each size class, temporal and spatial scheme. Further particulars of these estimators can be found in Link and Almeida (2000).

The approach to calculate consumption followed previously established and described methods for estimating consumption, using an evacuation rate model methodology. For further details, see Durbin et al. (1983), Ursin et al. (1985), Pennington (1985), Overholtz et al. (1991, 1999, 2000, 2008), Tsou & Collie (2001a, 2001b), Link & Garrison (2002), Link et al. (2002, 2006, 2008, 2009), Methratta & Link (2006), Link & Sosebee (2008), Overholtz & Link (2007, 2009), Tyrrell et al. (2007, 2008), Link and Idoine (2009), Moustahfid et al. (2009a, 2009b), and NEFSC (2006, 2007a, 2007b, 2008). The main data inputs are mean stomach contents (S_i) for each pollock size-time-space scheme i, diet composition (D_{ij}) where j is the specific prey of interest, and T is the bottom temperature taken from the bottom trawl surveys (Taylor et al. 2005). Estimates of variance about all these variables (data inputs) were calculated. Further particulars of these estimators can be found in Link and Almeida (2000). Again, units for stomach estimates are in g.

More specifically, using the evacuation rate model to calculate consumption requires two variables and two parameters. The per capita consumption rate, C_i is calculated as:

$$C_i = 24 \cdot E_i \cdot \overline{S_i}^{\gamma} \quad ,$$

where 24 is the number of hours in a day and the evacuation rate E_i is:

$$E_i = \alpha e^{\beta T} \quad ;$$

and is formulated such that estimates of mean stomach contents (S_i) and ambient temperature (T; here used as bottom temperature from the NEFSC bottom trawl surveys (Taylor et al. 2005)) are the only data required. The parameters • and • are set as values chosen from the literature (Tsou and Collie 2001a, 2001b, Overholtz 1999, 2000). The parameter • is a shape function is almost always set to 1 (Gerking 1994). As noted, to estimate per capita consumption, the gastric evacuation rate method was used (Eggers 1977, Elliott and Persson 1978). There has been copious experience in this region using these models (see references listed above). The two main parameters, • and •, were set to 0.004 and 0.11 respectively based upon prior studies and sensitivity analyses (NEFSC 2007a, 2007b). From 1992 and forward (when individual weights were measured), a diagnostic of % daily ration was also calculated.

Once per capita consumption rates were estimated for each pollock size class, temporal and spatial scheme, those estimates were then scaled up to an annual and stock wide basis, *C*:

$$C = 365 \cdot C_i \cdot N_i$$

where N_i is the estimate of abundance (see stock assessment results) for each pollock size class, temporal and spatial scheme and 365 is the number of days in a year.

This total consumption was partitioned for the major prey items of pollock by multiplying it by the diet composition of each prey (D_{ij}) to provide an estimate of prey removals. Both the total consumption and the amount of prey removed by each pollock size class (and combined across sizes) are presented as metric tons year^{-1}.

To evaluate the consumptive demands of a pollock and the predatory removals of pollock in a broader ecosystem context, two contrasts were executed. First, comparisons of total consumption by pollock were compared to the amount of energy flows for the entire ecosystem. These total energy flows were calculated in a recent energy budget (Link et al. 2006, 2008, 2009). Pollock consumption is presented as a percentage of total energy flows in the ecosystem.

Second, the total amount of commercially targeted prey eaten by pollock was treated as a removal. These estimates were then compared to concurrently estimated fishery landings to provide an evaluation of potential competition between pollock and fisheries on some of their major prey.

Results and Observations:

- From recent energy budgets, the amount of food consumed by pollock is 0.001-0.007% of all energy flows in the system.
- From recent energy budgets, pollock comprise 0.5-5% of the total consumption by all finfish on GB & GoM.
- This has changed over time, mainly as a function of pollock abundance.
- All diagnostics were within the normal range.
- Pollock consumption has been more important at times, perhaps when other piscivore species were at lower abundances, but has never been the dominant piscivore.

Summary:

- Abundance, landings, consumption, energy flow, and relative importance to overall system peaked in late 1990s to early 2000s (Figure C65).
- Trends are similar to prior studies (Tyrrell et al. 2007).
- These estimates are 1-2 orders of magnitude lower than other, previous estimates: mainly due to a more conservative choice of the • parameter.

- Pollock remain an ecologically important piscivore and shrimpivore in the NEUS ecosystem.
- Pollock probably do not consume a significant amount of certain species (relative to those spp. B, P, F), except for pandalid shrimp and maybe herring.

Research Recommendations

Term of Reference 8: Research Recommendations
Review, evaluate and report on the status of the SARC and Working Group research recommendations listed in recent SARC reviewed assessments and review panel reports. Identify new research recommendations.

The WG offers several research recommendations, prioritized below.

- Selectivity studies
    - Physical selectivity (e.g., multi-mesh gillnet)
    - Behavioral studies (e.g., swimming endurance, escape behavior)
    - Explore geographic and vertical distribution by size and age
    - Tag-recovery at size or age
    - Evaluate information on length-specific selectivity at older ages
- Stock definition – sensitive genetic markers
- Alternative pollock surveys (fixed gear, etc.)
- Examine how to incorporate Bigelow survey given that no calibration is available
- Explore inclusion of existing surveys (e.g., age composition of summer survey, inshore recruitment indices)
- Consider new survey approaches, because trawls surveys don’t survey pollock well (off-bottom, hard-bottom, fast-swimmers, patchy, …)
- Further evaluate age determination of old fish
- Investigate magnitude of historical discards
- Discard mortality studies (by gear)
- This assessment uses relative estimates (stratified mean) for survey indices. Investigating area swept estimates could be a research recommendation for the future.
- Investigating the use of party charter logbooks for recreational catch-at-age could be considered as a research recommendation.

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Tables

Table C1. Regulations summary

General Provisions			
Open Access		1977-1993	
Limited Entry		1994 -	
Days-at-sea Limits		1994-1996	Some groundfish vessels
		1996-2009	Almost all groundfish vessels
		2010-	Some groundfish vessels
Quotas		1977-1981	Cod, haddock, yellowtail only
		2004-2009	GB yellowtail flounder; portions of GB cod and haddock
		2010-	Sector vessels, most stocks
Small-mesh fishery provisions		1981-	Various programs
Mesh Size			
Gear	*Area*	*Years*	*Size*
Trawl	GOM/GB	1977-1981	4 ½" body/ 5 1/8" cod end
		1982	5 1/8"
		1983 – 1993	5 ½" throughout net
		1994-1997	6" (A5)
		1999-2000	6 ½" square, 6" diamond codend (FW 27)
		2002-	6 ½" square or diamond codend
	SNE/MA	1994-1998	6"
		199-2001	6 ½" sq, 6" dia.
		2002-	6 ½" sq. or dia.
Sink Gillnet	GOM/GB	1982-1985	5 ½"
	GOM/GB/SNE/MA	1986-1993	5 ½"
		1994-2001	6"
		2002-	6 ½"
Closures			
	CAI	1977-1994	Seasonal
		1995-	Year round
	CAII	1977-1994	Seasonal
		1995-	Year round
	SNE	1986-1993	Seasonal
	NLCA	1994	Seasonal
		1995-	Year round
	WGOM	1998-	Year round
	Cashes Ledge	1998-2001	Seasonal
		2002--	Year round
	GOM Rolling	1998-	Seasonal
	GB May	2000-	Seasonal

Table C2. Total catch (mt) of pollock in US areas 5&6 by commercial and recreational fisheries.

Year	US Landings	US Discards	Canadian Landings	Distant Water Fleet Landings	**Commercial Total mt**	Recreational Landings	Recreational Discards	**Recreational Total mt**	Total Catch (mt)
1960	8190	0	2211	0	10401	0	0	0	10401
1961	7861	0	359	0	8220	0	0	0	8220
1962	5550	0	601	0	6151	0	0	0	6151
1963	4673	0	953	615	6241	0	0	0	6241
1964	4764	0	1942	2298	9004	0	0	0	9004
1965	4903	0	2044	2040	8987	0	0	0	8987
1966	3232	0	4012	2664	9908	0	0	0	9908
1967	2741	0	5287	449	8477	0	0	0	8477
1968	2913	0	1740	499	5152	0	0	0	5152
1969	3521	0	2443	3872	9836	0	0	0	9836
1970	3586	0	853	7116	11555	0	0	0	11555
1971	4734	0	1636	7949	14319	0	0	0	14319
1972	5248	0	1366	6381	12995	0	0	0	12995
1973	5753	0	1727	5600	13080	0	0	0	13080
1974	7720	0	3539	755	12014	0	0	0	12014
1975	8190	0	4736	556	13482	0	0	0	13482
1976	9593	0	2116	1022	12731	0	0	0	12731
1977	11999	0	3413	104	15516	0	0	0	15516
1978	16758	0	4754	0	21512	0	0	0	21512
1979	14613	0	3032	0	17645	0	0	0	17645
1980	16567	0	5634	0	22201	0	0	0	22201
1981	17766	0	4050	0	21816	752	407	1159	22975
1982	13961	0	5373	1	19335	819	755	1573	20909
1983	13842	0	4383	0	18225	581	733	1313	19539
1984	17657	0	3290	0	20947	115	65	180	21126
1985	19192	0	1764	0	20956	259	58	317	21273
1986	24339	0	654	1	24994	143	34	177	25171
1987	20251	0	0	0	20251	115	187	303	20554
1988	14830	0	0	0	14830	167	406	573	15403
1989	10553	473	0	0	11025	259	236	496	11521
1990	9559	107	0	0	9666	155	116	271	9937
1991	7886	223	0	0	8109	100	289	389	8498
1992	7184	196	0	0	7380	50	47	97	7477
1993	5674	100	0	0	5774	52	58	110	5884
1994	3763	154	0	0	3918	253	202	455	4373
1995	3352	192	0	0	3544	247	514	761	4305
1996	2962	230	0	0	3192	339	223	562	3754
1997	4264	124	0	0	4388	196	172	368	4756
1998	5572	68	0	0	5640	128	186	314	5954
1999	4590	141	0	0	4730	89	141	230	4961
2000	4043	117	0	0	4160	243	356	599	4759
2001	4109	73	0	0	4182	471	875	1346	5528
2002	3580	68	0	0	3648	547	613	1160	4808
2003	4794	45	0	0	4839	499	472	971	5810

Table C2 (cont).

Year	US Landings	US Discards	Canadian Landings	Distant Water Fleet Landings	**Commercial Total mt**	Recreational Landings	Recreational Discards	**Recreational Total mt**	Total Catch (mt)
2004	5070	103	0	0	5173	669	241	910	6083
2005	6509	100	0	0	6609	520	272	792	7401
2006	6067	69	0	0	6136	571	252	823	6959
2007	8372	147	0	0	8518	533	227	760	9278
2008	9965	362	0	0	10327	941	926	1867	12194
2009	7477	*362*	0	0	7839	468	428	896	8735

Table C3. Port samples (sampling intensity) for pollock.

Year	Number of Fish Lengths	Number of Aged Fish	Commcial Landings (mt)	Lengths per mt	Ages per mt
1970	396	---	3586	0.11	---
1971	57	---	4734	0.01	---
1972	633	---	5248	0.12	---
1973	965	---	5753	0.17	---
1974	1053	---	7720	0.14	---
1975	548	---	8190	0.07	---
1976	497	60	9593	0.05	0.01
1977	4695	1099	11999	0.39	0.09
1978	2159	451	16758	0.13	0.03
1979	5716	1365	14613	0.39	0.09
1980	2412	548	16567	0.15	0.03
1981	5448	1346	17766	0.31	0.08
1982	5809	1314	13961	0.42	0.09
1983	9616	2415	13842	0.69	0.17
1984	7605	1811	17657	0.43	0.10
1985	7900	2050	19192	0.41	0.11
1986	9515	2438	24339	0.39	0.10
1987	8128	2162	20251	0.40	0.11
1988	9067	2128	14830	0.61	0.14
1989	7954	1853	10553	0.75	0.18
1990	6179	1429	9559	0.65	0.15
1991	6089	1418	7886	0.77	0.18
1992	6071	1405	7184	0.85	0.20
1993	4733	737	5674	0.83	0.13
1994	4466	1121	3763	1.19	0.30
1995	3043	753	3352	0.91	0.22
1996	3879	889	2962	1.31	0.30
1997	6738	1574	4264	1.58	0.37
1998	3198	822	5572	0.57	0.15
1999	4134	1168	4590	0.90	0.25
2000	3617	1006	4043	0.89	0.25
2001	5087	1385	4109	1.24	0.34
2002	3240	1133	3580	0.91	0.32
2003	9719	3360	4794	2.03	0.70
2004	8996	1640	5070	1.77	0.32
2005	7599	1598	6509	1.17	0.25
2006	8396	1985	6067	1.38	0.33
2007	7606	1802	8372	0.91	0.22
2008	7607	1558	9965	0.76	0.16
2009	8190	1612	7477	1.10	0.22

Table C4. Discards (mt) by fleet and NAFO area (in US waters of areas 5&6).

	Area 5				**Area 6**				
YEAR	Otter Trawl (large mesh)	Otter Trawl (small mesh)	Gillnet (large mesh)	Gillnet (x-large mesh)	Otter Trawl (large mesh)	Otter Trawl (small mesh)	Gillnet (large mesh)	Gillnet (x-large mesh)	Total Discards (all gears and areas)
1989	467.3	5.3	0.0	0.0	0.0	0.0	0.0	0.0	473
1990	103.3	3.9	0.0	0.0	0.0	0.0	0.0	0.0	107
1991	222.7	0.5	0.0	0.0	0.0	0.0	0.0	0.0	223
1992	194.9	0.8	0.0	0.0	0.0	0.0	0.0	0.0	196
1993	91.6	8.7	0.0	0.0	0.0	0.0	0.0	0.0	100
1994	17.0	4.9	131.7	0.6	0.0	0.0	0.0	0.0	154
1995	46.3	1.2	144.3	0.5	0.0	0.0	0.0	0.0	192
1996	54.4	45.5	129.3	0.6	0.0	0.1	0.0	0.0	230
1997	22.2	26.4	74.7	0.2	0.0	0.0	0.0	0.0	124
1998	5.5	7.2	54.9	0.5	0.0	0.0	0.0	0.0	68
1999	3.5	45.2	90.0	2.2	0.0	0.0	0.0	0.0	141
2000	28.0	6.2	79.7	3.2	0.0	0.0	0.0	0.0	117
2001	16.1	1.4	52.2	3.8	0.0	0.0	0.0	0.0	73
2002	9.8	0.8	56.3	1.3	0.0	0.0	0.0	0.0	68
2003	14.7	0.6	27.2	1.9	0.0	0.1	0.0	0.0	45
2004	41.2	2.2	51.2	6.3	1.8	0.0	0.0	0.0	103
2005	28.3	5.9	56.4	9.1	0.0	0.0	0.0	0.0	100
2006	10.5	0.1	51.1	7.6	0.0	0.0	0.0	0.0	69
2007	19.7	3.6	122.1	1.3	0.0	0.0	0.0	0.0	147
2008	16.1	8.8	333.0	3.8	0.0	0.0	0.0	0.0	362

Table C5. Survey attributes. The years where age structure is available pertains to pollock specifically (some age information is available earlier in the time series for other stocks).

Survey	Index	Years	Precision	%tows>0	Area	depth (m)	speed (kn)	duration(min)	height (m)	changes	comments
Fall	abundance	1963-2008(9)	CV~40%	0.24	GOM-GB	>30	3.8	30	1-2	D85, V~	
	age structure	1970-2008(9)									
Spring	abundance	1968-2008(9)	CV~30%	0.29	GOM-GB	>30	2	30	1-2	D85, N73-81,V~	
	age structure	1970-2008(9)									
Shrimp	abundance	1985-2009	CV~50%	0.36	W.GOM	?	3.8	15	3	none	no ages
Larval	SSB	1977-2008	IQR~?		SW.GOM-GB	>30	N/A		N/A	mesh93	
ME-NH	recruitment	2000-2009	?		inshore ME	<30	2.5	20	3	none	no ages
MAspring	recruitment	(1978)1982-2009	?	0.06	Inshore MA			15	3	V82	intermittent ages
MAfall	recruitment	(1978)1982-2009	?	0.036	inshore MA	<100~	2	15	3		intermittent ages

Table C6a. NEFSC spring survey age structure for pollock.

			N/tow at age											
Year	N/tow	CV	1	2	3	4	5	6	7	8	9	10	11	12+
1970	1.09	0.24	0.076	0.038	0.118	0.065	0.036	0.066	0.098	0.177	0.057	0.050	0.042	0.270
1971	0.80	0.18	0.035	0.092	0.131	0.080	0.060	0.063	0.008	0.054	0.012	0.044	0.044	0.176
1972	3.38	0.50	0.528	1.597	0.650	0.026	0.061	0.019	0.054	0.117	0.050	0.071	0.013	0.189
1973	4.56	0.45	0.006	3.293	0.589	0.167	0.125	0.026	0.015	0.090	0.015	0.150	0.010	0.078
1974	1.34	0.25	0.000	0.065	0.569	0.163	0.056	0.143	0.066	0.022	0.000	0.022	0.105	0.132
1975	1.43	0.31	0.000	0.232	0.172	0.335	0.039	0.073	0.086	0.082	0.036	0.065	0.019	0.288
1976	1.69	0.19	0.049	0.100	0.166	0.171	0.255	0.113	0.172	0.174	0.127	0.033	0.054	0.273
1977	1.61	0.32	0.108	0.475	0.219	0.065	0.151	0.274	0.143	0.104	0.012	0.005	0.004	0.047
1978	1.94	0.50	0.000	0.270	0.413	0.515	0.314	0.116	0.087	0.047	0.076	0.037	0.022	0.045
1979	0.95	0.19	0.111	0.051	0.084	0.072	0.135	0.104	0.062	0.138	0.069	0.025	0.030	0.065
1980	1.43	0.31	0.099	0.181	0.093	0.293	0.248	0.154	0.236	0.055	0.027	0.007	0.000	0.033
1981	1.43	0.25	0.006	0.375	0.049	0.072	0.163	0.209	0.070	0.061	0.052	0.089	0.055	0.227
1982	3.96	0.46	0.107	1.514	0.855	0.733	0.122	0.267	0.113	0.116	0.045	0.000	0.030	0.059
1983	0.88	0.33	0.570	0.059	0.019	0.029	0.002	0.000	0.048	0.026	0.008	0.012	0.017	0.088
1984	1.03	0.27	0.171	0.128	0.115	0.122	0.115	0.102	0.045	0.038	0.036	0.039	0.039	0.076
1985	15.20	0.85	0.015	0.336	4.445	3.591	4.545	1.774	0.243	0.017	0.068	0.064	0.006	0.091
1986	1.88	0.42	0.049	0.149	0.067	0.197	0.102	0.417	0.381	0.130	0.071	0.026	0.108	0.184
1987	1.66	0.68	0.153	0.908	0.201	0.025	0.035	0.036	0.074	0.080	0.050	0.006	0.018	0.070
1988	0.78	0.23	0.402	0.024	0.078	0.014	0.000	0.031	0.022	0.056	0.042	0.038	0.030	0.042
1989	1.90	0.50	0.057	0.124	0.105	0.437	0.408	0.283	0.170	0.144	0.034	0.069	0.000	0.070
1990	0.65	0.34	0.000	0.024	0.238	0.092	0.032	0.051	0.041	0.033	0.041	0.026	0.022	0.044
1991	2.05	0.26	0.110	0.076	0.434	0.589	0.310	0.258	0.158	0.011	0.048	0.009	0.025	0.025
1992	1.75	0.30	0.715	0.195	0.146	0.141	0.165	0.082	0.090	0.038	0.011	0.029	0.075	0.067
1993	1.62	0.34	0.588	0.277	0.327	0.196	0.046	0.089	0.048	0.014	0.011	0.017	0.008	0.000
1994	0.58	0.20	0.003	0.046	0.099	0.128	0.075	0.071	0.086	0.048	0.007	0.012	0.003	0.003
1995	3.58	0.83	0.004	0.022	0.868	1.974	0.512	0.124	0.003	0.049	0.012	0.012	0.000	0.000
1996	0.64	0.43	0.237	0.021	0.008	0.070	0.153	0.082	0.044	0.021	0.000	0.000	0.000	0.000
1997	3.54	0.40	0.513	0.478	0.776	0.593	0.712	0.193	0.193	0.034	0.031	0.013	0.000	0.000
1998	2.66	0.37	0.755	0.260	0.974	0.179	0.058	0.172	0.161	0.069	0.030	0.000	0.000	0.000
1999	2.22	0.45	0.653	1.115	0.181	0.130	0.038	0.051	0.042	0.012	0.000	0.000	0.000	0.000
2000	1.40	0.38	0.736	0.106	0.118	0.084	0.154	0.107	0.055	0.028	0.015	0.000	0.000	0.000
2001	1.72	0.31	0.671	0.166	0.119	0.075	0.257	0.245	0.115	0.050	0.000	0.013	0.000	0.005
2002	0.72	0.28	0.040	0.021	0.039	0.219	0.146	0.183	0.057	0.017	0.000	0.000	0.000	0.000
2003	1.44	0.69	0.303	0.861	0.046	0.074	0.038	0.052	0.040	0.016	0.000	0.000	0.013	0.000
2004	0.47	0.40	0.067	0.194	0.046	0.009	0.030	0.063	0.029	0.012	0.000	0.023	0.000	0.000
2005	2.17	0.38	0.006	0.454	0.015	0.031	0.136	0.932	0.375	0.155	0.043	0.020	0.000	0.000
2006	0.94	0.25	0.086	0.019	0.022	0.007	0.055	0.312	0.380	0.051	0.006	0.006	0.000	0.000
2007	2.09	0.24	0.235	0.141	0.203	0.087	0.318	0.426	0.662	0.023	0.000	0.000	0.000	0.000
2008	2.04	0.23	0.099	0.023	0.006	0.061	0.205	0.253	0.736	0.247	0.289	0.086	0.029	0.008
2009	1.00	0.26	0.140	0.218	0.145	0.011	0.091	0.049	0.032	0.205	0.063	0.019	0.025	0.000

Table C6b. NEFSC fall survey age structure for pollock.

			N/tow at age											
Year	N/tow	CV	1	2	3	4	5	6	7	8	9	10	11	12+
1970	0.55	0.20	0.071	0.089	0.006	0.105	0.092	0.069	0.045	0.029	0.010	0.012	0.010	0.013
1971	0.95	0.43	0.018	0.353	0.172	0.016	0.042	0.112	0.018	0.068	0.038	0.011	0.008	0.093
1972	1.48	0.26	0.343	0.294	0.210	0.092	0.079	0.093	0.084	0.075	0.053	0.026	0.036	0.098
1973	0.97	0.21	0.012	0.250	0.076	0.049	0.083	0.070	0.075	0.084	0.000	0.137	0.011	0.121
1974	0.99	0.35	0.002	0.078	0.322	0.235	0.097	0.085	0.112	0.000	0.014	0.000	0.031	0.030
1975	0.70	0.38	0.240	0.039	0.034	0.121	0.069	0.048	0.082	0.016	0.018	0.018	0.002	0.016
1976	4.30	0.48	0.038	0.032	0.169	0.580	1.938	0.651	0.350	0.210	0.054	0.008	0.000	0.266
1977	2.34	0.31	0.051	0.227	0.276	0.277	0.504	0.395	0.227	0.081	0.103	0.028	0.000	0.171
1978	1.07	0.21	0.033	0.221	0.044	0.051	0.110	0.082	0.172	0.081	0.070	0.039	0.024	0.140
1979	0.88	0.19	0.013	0.017	0.183	0.146	0.081	0.094	0.071	0.087	0.061	0.040	0.012	0.071
1980	0.49	0.21	0.057	0.006	0.011	0.049	0.096	0.031	0.047	0.049	0.019	0.056	0.023	0.049
1981	1.10	0.68	0.026	0.177	0.515	0.137	0.129	0.032	0.026	0.003	0.000	0.000	0.000	0.055
1982	0.79	0.36	0.082	0.221	0.222	0.053	0.018	0.057	0.048	0.000	0.024	0.000	0.017	0.050
1983	1.00	0.44	0.506	0.015	0.070	0.041	0.070	0.016	0.057	0.078	0.033	0.018	0.023	0.073
1984	0.28	0.36	0.104	0.123	0.017	0.004	0.003	0.020	0.003	0.003	0.005	0.000	0.000	0.000
1985	1.11	0.35	0.670	0.048	0.103	0.079	0.080	0.050	0.023	0.000	0.000	0.009	0.013	0.032
1986	0.42	0.28	0.135	0.082	0.032	0.039	0.042	0.043	0.038	0.008	0.000	0.000	0.005	0.000
1987	0.54	0.30	0.042	0.191	0.056	0.000	0.059	0.016	0.067	0.031	0.059	0.000	0.009	0.012
1988	3.96	0.66	0.096	0.116	1.106	1.351	0.432	0.449	0.079	0.192	0.085	0.020	0.008	0.028
1989	1.64	0.63	0.437	0.678	0.364	0.132	0.000	0.000	0.000	0.000	0.000	0.000	0.013	0.018
1990	0.77	0.33	0.010	0.089	0.246	0.151	0.124	0.009	0.022	0.034	0.023	0.038	0.000	0.026
1991	0.70	0.40	0.138	0.066	0.154	0.230	0.056	0.043	0.012	0.000	0.000	0.000	0.000	0.000
1992	0.91	0.53	0.303	0.200	0.132	0.131	0.113	0.016	0.010	0.000	0.000	0.000	0.000	0.000
1993	1.10	0.49	0.484	0.399	0.092	0.032	0.012	0.061	0.000	0.000	0.000	0.000	0.000	0.015
1994	0.37	0.37	0.000	0.051	0.137	0.098	0.071	0.018	0.000	0.000	0.000	0.000	0.000	0.000
1995	0.86	0.41	0.031	0.157	0.470	0.110	0.069	0.024	0.000	0.000	0.000	0.000	0.000	0.000
1996	1.01	0.40	0.288	0.309	0.046	0.212	0.134	0.015	0.006	0.000	0.000	0.000	0.000	0.000
1997	1.70	0.54	0.549	0.634	0.146	0.170	0.172	0.033	0.000	0.000	0.000	0.000	0.000	0.000
1998	2.07	0.66	1.243	0.328	0.319	0.092	0.028	0.035	0.022	0.000	0.000	0.000	0.000	0.000
1999	2.30	0.32	0.510	0.539	0.204	0.517	0.267	0.200	0.044	0.014	0.000	0.000	0.000	0.000
2000	2.45	0.74	0.350	1.949	0.093	0.017	0.027	0.018	0.000	0.000	0.000	0.000	0.000	0.000
2001	2.14	0.32	0.116	0.612	0.482	0.501	0.272	0.093	0.052	0.013	0.000	0.000	0.000	0.000
2002	3.18	0.43	0.203	0.131	0.923	0.691	0.830	0.326	0.075	0.000	0.000	0.000	0.000	0.000
2003	7.97	0.66	0.313	2.034	1.909	3.106	0.530	0.075	0.000	0.000	0.000	0.000	0.000	0.000
2004	3.11	0.55	0.116	0.260	1.661	0.418	0.361	0.203	0.087	0.000	0.000	0.000	0.000	0.000
2005	5.09	0.41	0.033	2.228	0.407	0.904	0.631	0.765	0.114	0.012	0.000	0.000	0.000	0.000
2006	1.68	0.66	0.282	0.803	0.115	0.052	0.102	0.155	0.168	0.006	0.000	0.000	0.000	0.000
2007	0.33	0.26	0.112	0.012	0.000	0.028	0.015	0.077	0.056	0.014	0.017	0.000	0.000	0.000
2008	1.01	0.57	0.153	0.262	0.231	0.080	0.044	0.026	0.048	0.048	0.047	0.035	0.016	0.022
2009	0.23	0.31	0.082	0.119	0.012	0.006	0.006	0.000	0.000	0.005	0.000	0.000	0.000	0.000

Table C7a. Commercial catch at age (in thousands of fish) of pollock in US waters of NAFO areas 5 and 6. In 2009, discards at age were not estimated and the amount of total discards was assumed to be equal to the 2008 amount.

Year	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9+
1970	0	645	436	990	884	563	392	243	213.1
1971	0	1044	1487	1267	1019	796	276	117	6.1
1972	0	286	777	1013	746	331	173	39	270.1
1973	0	566	864	2715	1493	204	82	29	149.1
1974	0	87	2414	1110	968	411	127	70	86.1
1975	0	107	530	1871	809	791	337	95	114.1
1976	0	79	905	1234	1948	466	354	81	29.1
1977	0	23	471	1259	870	1058	400	297	378.1
1978	0	91	824	1056	1141	810	1085	373	695.1
1979	0	200	1553	2225	1311	635	278	293	288.1
1980	0	194	415	2040	2189	1355	653	218	357.1
1981	0	587	1545	697	2014	1140	603	322	411.1
1982	0	120	1616	894	366	1005	683	437	636.1
1983	0	36	1047	3252	814	222	428	283	623.1
1984	0	44	574	2172	3609	697	123	180	423.1
1985	0	196	1854	758	1794	2043	334	87	411.1
1986	0	54	940	3120	927	1650	1208	182	427.1
1987	0	81	950	856	2703	546	637	413	396.1
1988	0	0	360	803	848	1614	441	262	281.1
1989	53	111	321	1352	801	457	504	190	215
1990	13	13	645	911	1142	375	201	146	224
1991	152	66	186	798	610	664	164	77	194
1992	197	112	78	459	754	440	347	81	100
1993	413	40	108	136	320	546	273	148	63
1994	8	4	3	62	181	283	240	95	86
1995	21	12	30	107	174	233	208	86	54
1996	96	40	66	166	224	258	141	75	29
1997	1	9	24	160	451	366	193	75	44
1998	1	2	15	45	322	696	335	93	25
1999	1	12	23	171	253	402	326	107	44
2000	0	1	26	118	376	334	175	93	61
2001	0	2	31	162	292	399	222	90	66
2002	0	8	19	96	259	166	231	112	78
2003	0	5	7	101	290	373	221	165	106
2004	15	7	11	14	160	406	371	170	146
2005	2	3	7	31	70	538	618	283	149
2006	2	0	5	5	96	183	638	366	171
2007	3	2	11	52	82	572	379	620	350
2008	3	19	48	52	96	192	946	358	698
2009	0	0	15	122	83	272	274	477	575

Table C7b. Recreational catch at age (in thousands of fish) of pollock in US waters of NAFO areas 5 and 6.

Year	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9+
1970	0	0	0	0	0	0	0	0	0
1971	0	0	0	0	0	0	0	0	0
1972	0	0	0	0	0	0	0	0	0
1973	0	0	0	0	0	0	0	0	0
1974	0	0	0	0	0	0	0	0	0
1975	0	0	0	0	0	0	0	0	0
1976	0	0	0	0	0	0	0	0	0
1977	0	0	0	0	0	0	0	0	0
1978	0	0	0	0	0	0	0	0	0
1979	0	0	0	0	0	0	0	0	0
1980	0	0	0	0	0	0	0	0	0
1981	336	1473	222	28	96	31	5	3	3
1982	99	705	393	25	19	26	11	12	74
1983	274	63	214	95	6	2	2	1	101
1984	150	246	53	16	5	0	0	0	0
1985	506	331	202	49	74	51	17	11	66
1986	358	35	44	7	1	0	0	1	6
1987	329	281	29	0	8	1	0	0	4
1988	948	168	76	22	1	4	1	1	17
1989	119	207	67	134	21	4	2	2	24
1990	58	50	76	40	14	4	0	0	0
1991	186	126	18	44	18	2	0	2	4
1992	71	33	23	13	8	0	2	0	2
1993	101	177	104	8	7	0	0	0	0
1994	73	146	442	143	40	12	4	0	3
1995	221	123	273	154	27	6	2	0	1
1996	121	55	46	137	60	30	5	1	0
1997	19	71	36	66	67	14	8	2	0
1998	53	56	85	63	94	81	11	2	1
1999	244	196	14	38	30	20	14	1	1
2000	651	222	88	14	20	40	30	3	5
2001	9	430	253	102	52	108	69	33	3
2002	0	20	115	64	198	40	43	11	5
2003	0	56	14	35	92	96	31	18	15
2004	4	18	9	8	80	107	53	19	10
2005	1	8	10	31	26	75	66	24	13
2006	18	16	30	11	30	35	81	37	20
2007	1	5	12	47	18	55	35	44	22
2008	2	17	23	26	36	45	179	63	108
2009	2	12	14	23	9	28	35	43	74

Table C8. Estimated spawning biomass at age per year from the ASAP base model (reported to 3 significant digits). Spawning weights were calculated as January 1 weights by applying the Rivard method to mid-year catch weights.

Year	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9+	Total
1970	44	541	2150	6320	13500	11700	19700	37600	206000	297000
1971	34	616	3920	6910	12000	16800	12900	20900	253000	327000
1972	109	710	5650	14700	13700	16500	18600	13500	233000	316000
1973	29	995	5270	16800	20800	15400	15500	17900	162000	254000
1974	30	489	7980	12700	22700	20800	12500	12500	159000	248000
1975	37	521	3780	30100	22100	26600	22200	12400	153000	271000
1976	35	443	3930	10200	45500	25100	24900	19700	140000	270000
1977	34	617	3300	10100	16300	50600	23700	23700	128000	256000
1978	12	538	4610	10100	16300	19400	49900	22600	119000	243000
1979	22	164	3950	11500	15900	18500	18600	46200	129000	244000
1980	69	389	1220	10600	17900	17700	16400	16400	148000	229000
1981	73	591	2890	3790	15900	18200	14600	13900	135000	205000
1982	17	398	3680	7540	5800	16100	14900	12200	130000	191000
1983	55	206	2270	11700	10700	5270	13200	12400	124000	180000
1984	31	460	1410	8500	17900	8770	4040	10600	96800	148000
1985	14	203	2770	3840	12700	17400	6750	3270	96100	143000
1986	38	219	1220	8170	5100	11400	13600	5950	80400	126000
1987	14	306	1590	4420	12500	4160	7300	8230	66600	105000
1988	35	149	2110	4320	6270	10500	2540	4340	54200	84500
1989	22	247	1190	7140	6390	5280	7020	1550	48200	77100
1990	14	114	1680	4220	10800	6180	4030	5060	36500	68600
1991	18	78	780	6640	6650	11600	5190	3030	36600	70500
1992	39	170	591	3380	11700	7730	10500	4420	33600	72200
1993	44	237	960	2740	5430	13900	7480	9310	32600	72700
1994	27	238	1040	2780	4350	6430	13400	6780	37600	72600
1995	33	197	1680	4430	6470	5660	6490	12700	45700	83300
1996	50	299	1470	8500	13600	10200	6070	6160	44900	91300
1997	42	385	1740	5970	16100	18400	11200	5950	44100	104000
1998	74	226	2340	5720	10300	20500	18900	10900	42900	112000
1999	110	485	1490	9070	11500	12700	21500	18500	46300	122000
2000	105	504	2330	6800	18800	14800	13500	21100	52400	130000
2001	40	597	2440	9800	12600	23800	16700	13600	68200	148000
2002	42	265	4280	11500	25600	15800	25200	16200	67400	166000
2003	23	363	1560	19800	28200	32100	17100	25500	73900	199000
2004	22	108	2520	5990	33300	36100	33300	16200	86300	214000
2005	7	193	681	10500	12900	41800	38500	31800	85600	222000
2006	20	95	1260	3390	20900	18500	45900	38000	108000	236000
2007	14	183	941	5230	7730	28000	21800	44900	115000	224000
2008	29	161	1350	3950	11800	11700	31100	21000	146000	227000
2009	41	192	1400	5180	7610	15100	12300	26900	128000	196000

Table C9. Estimated numbers (thousands of fish) at age per year from the ASAP base model (reported to 3 significant digits).

Year	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9+	Total
1970	28700	19600	9550	7990	7460	4380	5230	7440	28300	119000
1971	27000	23500	15900	7420	5880	5420	3180	3950	28600	121000
1972	57500	22100	19000	12300	5370	4190	3850	2370	26100	153000
1973	20900	47100	17900	14800	9140	3950	3090	2940	23000	143000
1974	22000	17100	38100	13900	10800	6560	2840	2310	20800	134000
1975	22400	18000	13900	29900	10400	8050	4880	2180	18700	128000
1976	25800	18300	14600	10900	22600	7800	6010	3760	16800	127000
1977	23700	21100	14900	11500	8260	16900	5830	4630	16600	123000
1978	7620	19400	17100	11600	8520	6030	12300	4420	17000	104000
1979	16100	6240	15600	12900	8110	5830	4130	8960	17000	95000
1980	35300	13200	5040	11900	9190	5650	4060	3030	20600	108000
1981	24200	28900	10500	3700	7840	5870	3610	2820	18600	106000
1982	9980	19300	22600	7500	2330	4810	3610	2440	16800	89300
1983	25100	7840	14900	15900	4730	1440	2970	2450	15000	90400
1984	10700	19900	6070	10600	10100	2930	892	2020	13700	76800
1985	11200	8680	15700	4280	6430	5940	1720	585	12200	66800
1986	19200	9090	6820	11000	2550	3690	3410	1110	10000	66800
1987	11900	15300	7320	5130	6790	1340	1820	1680	8260	59600
1988	19000	9460	12300	5500	3190	3600	667	903	7260	61800
1989	8470	15000	7520	9230	3480	1740	1850	344	6100	53700
1990	7240	6690	11900	5740	6180	2080	998	1060	4960	46800
1991	11900	5790	5370	9230	4000	3930	1280	613	4600	46700
1992	19300	9550	4660	4210	6670	2700	2580	839	4080	54600
1993	22200	15600	7760	3700	3130	4680	1850	1770	3860	64600
1994	13600	18000	12700	6190	2800	2260	3330	1320	4410	64700
1995	15500	11100	14600	10300	4920	2110	1610	2380	4550	67000
1996	24300	12500	8950	11800	8160	3750	1530	1180	5500	77700
1997	16000	19800	10200	7260	9460	6340	2800	1150	5380	78400
1998	33200	13100	16100	8290	5840	7360	4740	2100	5250	95900
1999	41300	27100	10700	13100	6680	4540	5470	3530	5880	118000
2000	50300	33700	22100	8690	10600	5230	3430	4140	7520	146000
2001	22400	41000	27500	18000	7000	8330	3980	2620	9360	140000
2002	34700	18200	33200	22200	14400	5480	6350	3050	9680	147000
2003	13800	28400	14900	27100	18000	11400	4240	4910	10300	133000
2004	18100	11300	23200	12100	22000	14300	8860	3280	12200	125000
2005	11900	14800	9230	19000	9890	17800	11300	6830	12400	113000
2006	14000	9730	12100	7540	15500	8000	14000	8670	15300	105000
2007	16400	11400	7960	9870	6150	12500	6320	10800	19100	101000
2008	20800	13400	9360	6500	8040	4950	9740	4760	23600	101000
2009	20800	17000	10900	7620	5260	6390	3730	6970	22100	101000

Table C10a. Spawning weights at age, derived by applying the Rivard method to mid-year catch weights at age.

Year	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9+
1970	0.08	0.35	0.87	1.34	2.11	2.78	3.81	5.07	7.28
1971	0.06	0.34	0.95	1.57	2.37	3.23	4.11	5.29	8.83
1972	0.09	0.41	1.15	2.03	2.97	4.09	4.88	5.70	8.92
1973	0.07	0.27	1.13	1.92	2.65	4.06	5.07	6.10	7.05
1974	0.07	0.37	0.81	1.55	2.45	3.29	4.46	5.43	7.64
1975	0.08	0.37	1.05	1.70	2.47	3.44	4.59	5.72	8.23
1976	0.07	0.31	1.04	1.57	2.35	3.34	4.18	5.25	8.32
1977	0.07	0.38	0.86	1.48	2.30	3.11	4.10	5.12	7.70
1978	0.08	0.36	1.04	1.48	2.23	3.35	4.08	5.12	7.01
1979	0.07	0.34	0.97	1.50	2.29	3.30	4.54	5.17	7.56
1980	0.10	0.38	0.94	1.50	2.27	3.26	4.07	5.42	7.21
1981	0.15	0.26	1.06	1.73	2.36	3.23	4.08	4.95	7.28
1982	0.09	0.26	0.63	1.70	2.90	3.49	4.17	5.03	7.77
1983	0.11	0.34	0.59	1.24	2.64	3.81	4.50	5.06	8.24
1984	0.15	0.30	0.90	1.36	2.07	3.11	4.57	5.23	7.07
1985	0.06	0.30	0.68	1.51	2.30	3.05	3.96	5.61	7.86
1986	0.10	0.31	0.69	1.26	2.33	3.22	4.03	5.37	8.03
1987	0.06	0.26	0.84	1.46	2.14	3.22	4.06	4.91	8.07
1988	0.09	0.20	0.66	1.33	2.29	3.04	3.85	4.82	7.47
1989	0.13	0.21	0.61	1.31	2.14	3.15	3.83	4.51	7.91
1990	0.10	0.22	0.55	1.24	2.03	3.09	4.08	4.77	7.36
1991	0.07	0.17	0.56	1.22	1.94	3.06	4.10	4.96	7.94
1992	0.10	0.23	0.49	1.36	2.05	2.98	4.12	5.28	8.25
1993	0.10	0.19	0.48	1.25	2.02	3.09	4.08	5.27	8.44
1994	0.10	0.17	0.32	0.76	1.81	2.95	4.06	5.17	8.53
1995	0.11	0.23	0.44	0.73	1.53	2.78	4.06	5.33	10.05
1996	0.10	0.31	0.63	1.22	1.94	2.84	4.00	5.25	8.17
1997	0.13	0.25	0.66	1.39	1.99	3.02	4.05	5.19	8.20
1998	0.11	0.22	0.56	1.17	2.06	2.89	4.04	5.19	8.17
1999	0.13	0.23	0.54	1.17	2.01	2.92	3.97	5.25	7.89
2000	0.10	0.19	0.41	1.32	2.06	2.95	3.98	5.10	6.97
2001	0.09	0.19	0.34	0.92	2.10	2.97	4.22	5.19	7.29
2002	0.06	0.19	0.50	0.87	2.08	3.00	4.01	5.33	6.97
2003	0.08	0.16	0.41	1.23	1.83	2.92	4.08	5.22	7.20
2004	0.06	0.12	0.42	0.83	1.77	2.63	3.80	4.96	7.06
2005	0.03	0.17	0.28	0.93	1.52	2.45	3.45	4.67	6.90
2006	0.07	0.12	0.40	0.76	1.57	2.41	3.31	4.39	7.05
2007	0.04	0.20	0.46	0.89	1.47	2.33	3.48	4.15	6.01
2008	0.07	0.15	0.56	1.03	1.71	2.46	3.23	4.43	6.20
2009	0.10	0.14	0.49	1.15	1.69	2.45	3.33	3.87	5.77

Table C10b. Catch weights at age, assumed to reflect mid-year weights at age.

Year	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9+
1970	0.16	0.58	1.17	1.78	2.61	3.38	4.49	5.72	7.28
1971	0.16	0.71	1.56	2.12	3.16	4.00	4.99	6.24	8.83
1972	0.16	1.06	1.86	2.65	4.17	5.29	5.95	6.52	8.92
1973	0.16	0.46	1.21	1.98	2.65	3.96	4.86	6.25	7.05
1974	0.16	0.84	1.42	1.98	3.02	4.09	5.03	6.06	7.64
1975	0.16	0.86	1.31	2.04	3.07	3.92	5.14	6.51	8.23
1976	0.16	0.60	1.25	1.89	2.71	3.64	4.46	5.37	8.32
1977	0.16	0.88	1.22	1.75	2.80	3.58	4.62	5.88	7.70
1978	0.16	0.79	1.23	1.79	2.85	4.01	4.66	5.67	7.01
1979	0.16	0.71	1.20	1.83	2.94	3.82	5.15	5.73	7.56
1980	0.16	0.90	1.24	1.87	2.82	3.61	4.33	5.71	7.21
1981	0.20	0.43	1.24	2.42	2.98	3.70	4.61	5.67	7.28
1982	0.17	0.35	0.92	2.33	3.47	4.09	4.69	5.48	7.77
1983	0.18	0.67	0.99	1.66	2.98	4.19	4.95	5.45	8.24
1984	0.21	0.49	1.20	1.87	2.57	3.25	4.98	5.53	7.07
1985	0.14	0.43	0.94	1.91	2.84	3.61	4.83	6.31	7.86
1986	0.16	0.68	1.11	1.69	2.84	3.65	4.50	5.97	8.03
1987	0.11	0.41	1.03	1.91	2.71	3.66	4.51	5.35	8.07
1988	0.14	0.37	1.07	1.71	2.75	3.41	4.04	5.15	7.47
1989	0.17	0.32	1.01	1.60	2.69	3.61	4.30	5.04	7.91
1990	0.13	0.28	0.93	1.53	2.58	3.54	4.60	5.29	7.36
1991	0.13	0.23	1.12	1.59	2.46	3.64	4.76	5.35	7.94
1992	0.14	0.40	1.04	1.64	2.64	3.61	4.67	5.86	8.25
1993	0.13	0.27	0.57	1.51	2.50	3.61	4.62	5.95	8.44
1994	0.15	0.22	0.37	1.01	2.17	3.49	4.56	5.78	8.53
1995	0.18	0.35	0.89	1.44	2.33	3.57	4.73	6.22	10.05
1996	0.16	0.52	1.15	1.67	2.61	3.47	4.48	5.82	8.17
1997	0.17	0.39	0.83	1.68	2.36	3.50	4.73	6.01	8.20
1998	0.16	0.29	0.80	1.64	2.52	3.55	4.66	5.69	8.17
1999	0.16	0.33	1.00	1.70	2.46	3.38	4.44	5.92	7.89
2000	0.14	0.23	0.50	1.75	2.50	3.53	4.69	5.86	6.97
2001	0.13	0.25	0.51	1.70	2.52	3.53	5.05	5.74	7.29
2002	0.10	0.27	0.99	1.50	2.54	3.57	4.55	5.62	6.97
2003	0.10	0.27	0.61	1.54	2.23	3.35	4.66	5.98	7.20
2004	0.10	0.15	0.65	1.14	2.03	3.10	4.31	5.28	7.06
2005	0.06	0.28	0.54	1.34	2.02	2.95	3.83	5.06	6.90
2006	0.12	0.26	0.58	1.07	1.85	2.88	3.71	5.04	7.05
2007	0.08	0.35	0.80	1.38	2.01	2.93	4.20	4.65	6.01
2008	0.10	0.30	0.88	1.32	2.12	3.01	3.56	4.68	6.20
2009	0.12	0.21	0.81	1.50	2.16	2.84	3.69	4.21	5.77

Table C11. Estimated January 1 total biomass at age per year from the ASAP base model (reported to 3 significant digits). January 1 weights are the same as spawning weights and were calculated by applying the Rivard method to mid-year catch weights.

Year	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9+	Total
1970	2180	6950	8300	10700	15700	12200	19900	37700	206000	319000
1971	1680	7910	15100	11700	14000	17500	13000	20900	253000	354000
1972	5430	9110	21800	24900	16000	17100	18800	13500	233000	359000
1973	1460	12800	20300	28500	24200	16100	15600	17900	162000	299000
1974	1520	6270	30800	21500	26400	21600	12700	12600	159000	292000
1975	1850	6680	14600	50900	25700	27700	22400	12500	154000	316000
1976	1760	5680	15200	17200	53100	26100	25100	19800	140000	304000
1977	1710	7920	12700	17000	19000	52600	23900	23700	128000	286000
1978	579	6900	17800	17100	19000	20200	50400	22600	119000	274000
1979	1080	2100	15200	19400	18600	19200	18800	46300	129000	270000
1980	3450	4990	4730	17900	20900	18400	16500	16400	148000	252000
1981	3660	7580	11100	6400	18500	18900	14700	14000	135000	230000
1982	854	5110	14200	12800	6760	16800	15000	12300	130000	214000
1983	2740	2650	8740	19700	12500	5480	13400	12400	124000	201000
1984	1560	5900	5440	14400	20900	9120	4080	10600	96800	169000
1985	713	2610	10700	6480	14800	18100	6810	3280	96200	160000
1986	1920	2810	4710	13800	5950	11900	13700	5970	80400	141000
1987	713	3930	6120	7470	14500	4330	7370	8240	66700	119000
1988	1760	1910	8140	7300	7320	10900	2560	4350	54200	98500
1989	1120	3170	4600	12100	7460	5490	7090	1550	48200	90800
1990	708	1460	6490	7130	12600	6420	4070	5070	36500	80400
1991	883	1000	3010	11200	7750	12000	5240	3040	36600	80800
1992	1950	2180	2280	5710	13700	8040	10600	4430	33600	82500
1993	2220	3040	3700	4630	6330	14400	7550	9330	32600	83900
1994	1340	3050	4020	4700	5070	6690	13500	6800	37600	82800
1995	1640	2530	6460	7490	7540	5890	6550	12700	45700	96500
1996	2490	3840	5680	14400	15800	10700	6130	6170	45000	110000
1997	2080	4950	6700	10100	18800	19200	11300	5970	44100	123000
1998	3690	2900	9010	9670	12000	21300	19100	10900	42900	132000
1999	5510	6220	5740	15300	13400	13200	21700	18500	46300	146000
2000	5270	6470	8980	11500	21900	15400	13600	21100	52400	157000
2001	2020	7660	9410	16600	14700	24800	16800	13600	68300	174000
2002	2110	3400	16500	19400	29900	16400	25400	16300	67400	197000
2003	1130	4670	6030	33400	32900	33400	17300	25600	73900	228000
2004	1080	1380	9730	10100	38800	37600	33700	16300	86300	235000
2005	342	2470	2630	17700	15000	43500	38800	31900	85600	238000
2006	983	1220	4870	5730	24300	19300	46300	38100	108000	249000
2007	675	2340	3630	8830	9020	29100	22000	45000	115000	235000
2008	1440	2070	5190	6680	13700	12200	31400	21100	146000	240000
2009	2070	2460	5390	8750	8880	15700	12400	27000	128000	210000

Table C12. Estimated exploitable biomass at age per year from the ASAP base model (reported to 3 significant digits). Mid-year catch weights were multiplied by numbers at age, and the exploitable fraction was obtained by further multiplying by selectivity at age by year.

Year	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9+	Total
1970	0	1010	4910	12600	19500	14800	15700	16800	24500	110000
1971	0	1480	10900	13900	18600	21700	10600	9730	30100	117000
1972	0	2080	15500	28700	22400	22200	15300	6100	27700	140000
1973	0	1920	9540	26000	24200	15700	10000	7260	19300	114000
1974	0	1270	23800	24300	32600	26800	9530	5540	18900	143000
1975	0	1370	7990	54000	32000	31500	16800	5600	18300	168000
1976	0	975	8040	18300	61200	28400	17900	7970	16700	159000
1977	0	1650	7990	17800	23100	60500	18000	10800	15200	155000
1978	0	1360	9250	18400	24300	24200	38400	9900	14200	140000
1979	0	393	8250	21000	23800	22300	14200	20300	15300	126000
1980	0	1050	2740	19700	25900	20400	11700	6840	17700	106000
1981	430	2090	6350	8070	23400	21500	11000	6250	16300	95400
1982	247	1480	10700	15900	8100	19400	11100	5210	15800	88000
1983	548	1040	7390	24000	14100	5950	9700	5210	15000	83000
1984	34	994	3260	17500	26000	9510	2960	4410	11600	76300
1985	50	436	6740	7290	18300	21400	5530	1450	11500	72600
1986	147	214	1260	10100	6320	13500	15300	4820	11300	63000
1987	74	267	1290	5380	16100	4920	8180	6530	9410	52100
1988	209	221	2440	5230	7680	12300	2690	3370	7680	41800
1989	146	402	1520	8330	8210	6290	7930	1250	6860	40900
1990	78	125	2080	4890	14000	7370	4580	4070	5180	42300
1991	151	107	1180	8250	8620	14300	6050	2370	5200	46200
1992	163	176	842	3800	15400	9740	12000	3570	4750	50400
1993	199	228	793	3090	6830	16900	8520	7640	4610	48800
1994	129	313	491	1370	3530	7900	14600	4270	4030	36600
1995	210	364	1590	3500	6780	7550	7260	8200	4840	40300
1996	230	483	1020	4210	12300	13000	6610	3860	4840	46600
1997	113	404	625	2310	12500	22200	12900	3980	4850	59900
1998	151	137	712	2320	8100	26100	21700	6970	4790	71000
1999	163	283	532	3700	8990	15300	23900	12300	5190	70400
2000	448	620	1170	3350	15400	18500	15400	13600	5620	74200
2001	346	1510	2570	9100	11100	29400	18600	7880	6850	87400
2002	42	113	1710	5840	19500	18900	28900	10800	9660	95400
2003	13	132	379	6790	21400	37400	19800	18200	9940	114000
2004	20	29	577	1110	9200	27400	38200	14700	22100	113000
2005	7	60	169	1850	3910	32000	43200	29400	21900	133000
2006	18	40	256	622	5770	14200	51900	37100	27600	138000
2007	12	52	198	922	2330	22300	26600	43000	29400	125000
2008	25	76	344	742	3670	9300	34700	18800	37400	105000
2009	25	52	300	830	2220	11100	13800	25000	32700	86000

Table C13. Estimated total pollock fishing mortality at age (both fleets combined), and the unweighted average F for ages 5 to 7 from the ASAP base model.

Year	Age 1	Age 2	Age 3	Age 4	Age 5	Age 6	Age 7	Age 8	Age 9+	Ave 5-7
1970	0	0.01	0.05	0.11	0.12	0.12	0.08	0.05	0.01	0.11
1971	0	0.01	0.06	0.12	0.14	0.14	0.09	0.06	0.02	0.12
1972	0	0.01	0.05	0.09	0.11	0.11	0.07	0.04	0.01	0.10
1973	0	0.01	0.06	0.12	0.13	0.13	0.09	0.05	0.02	0.12
1974	0	0.01	0.04	0.08	0.10	0.10	0.06	0.04	0.01	0.09
1975	0	0.01	0.04	0.08	0.09	0.09	0.06	0.04	0.01	0.08
1976	0	0.01	0.04	0.08	0.09	0.09	0.06	0.04	0.01	0.08
1977	0	0.01	0.05	0.10	0.11	0.11	0.08	0.05	0.01	0.10
1978	0	0.02	0.08	0.16	0.18	0.18	0.12	0.07	0.02	0.16
1979	0	0.01	0.07	0.14	0.16	0.16	0.11	0.06	0.02	0.14
1980	0	0.02	0.11	0.22	0.25	0.25	0.17	0.10	0.03	0.22
1981	0.03	0.05	0.14	0.26	0.29	0.29	0.19	0.11	0.04	0.26
1982	0.04	0.06	0.15	0.26	0.29	0.28	0.19	0.11	0.04	0.25
1983	0.03	0.06	0.14	0.25	0.28	0.28	0.18	0.11	0.03	0.25
1984	0.01	0.03	0.15	0.30	0.33	0.33	0.22	0.13	0.04	0.29
1985	0.01	0.04	0.16	0.32	0.36	0.35	0.24	0.14	0.04	0.32
1986	0.02	0.02	0.08	0.28	0.44	0.51	0.51	0.37	0.07	0.49
1987	0.03	0.02	0.09	0.27	0.44	0.50	0.50	0.36	0.07	0.48
1988	0.04	0.03	0.09	0.26	0.41	0.46	0.46	0.34	0.07	0.44
1989	0.04	0.03	0.07	0.20	0.31	0.36	0.36	0.26	0.05	0.34
1990	0.02	0.02	0.05	0.16	0.25	0.29	0.29	0.21	0.04	0.28
1991	0.02	0.02	0.04	0.12	0.19	0.22	0.22	0.16	0.03	0.21
1992	0.01	0.01	0.03	0.10	0.15	0.18	0.18	0.13	0.03	0.17
1993	0.01	0.01	0.03	0.08	0.12	0.14	0.14	0.10	0.02	0.13
1994	0.01	0.01	0.02	0.03	0.08	0.14	0.13	0.08	0.02	0.12
1995	0.01	0.01	0.02	0.03	0.07	0.12	0.12	0.07	0.01	0.10
1996	0.01	0.01	0.01	0.02	0.05	0.09	0.09	0.05	0.01	0.08
1997	0	0	0.01	0.02	0.05	0.09	0.09	0.05	0.01	0.08
1998	0	0	0.01	0.02	0.05	0.10	0.10	0.06	0.01	0.08
1999	0	0	0	0.01	0.04	0.08	0.08	0.05	0.01	0.07
2000	0	0.01	0.01	0.02	0.04	0.07	0.07	0.04	0.01	0.06
2001	0.01	0.01	0.01	0.02	0.05	0.07	0.07	0.04	0.01	0.06
2002	0	0	0	0.01	0.03	0.06	0.06	0.04	0.01	0.05
2003	0	0	0	0.01	0.03	0.06	0.06	0.04	0.01	0.05
2004	0	0	0	0	0.01	0.04	0.06	0.05	0.02	0.04
2005	0	0	0	0	0.01	0.04	0.06	0.05	0.02	0.04
2006	0	0	0	0	0.01	0.04	0.06	0.05	0.01	0.04
2007	0	0	0	0.01	0.02	0.05	0.08	0.07	0.02	0.05
2008	0	0	0.01	0.01	0.03	0.08	0.13	0.11	0.03	0.08
2009	0	0	0	0.01	0.02	0.07	0.12	0.10	0.03	0.07

Table C14. Model results for the ASAP base pollock model and several sensitivity models where the value for fixed selectivity at age 9+ in the indices was varied between 1.0 and 0.1. The model "Est Index.sel(9+)" allowed selectivity for the 9+ group to be freely estimated (estimates were 0.25 for spring and 0.22 for fall). SSB0 is unexploited spawning biomass. The shaded column is a sensitivity run including Canadian landings in area 5Zc (northeast corner of Georges Bank). Because it contains different data, likelihood components cannot be directly compared with the other models.

Model estimate	ASAP base model	Index.sel(9+)=1.0	Index.sel(9+)=0.9	Index.sel(9+)=0.8	Index.sel(9+)=0.7	Index.sel(9+)=0.6
lk.total	**4531**	4562	4562	4553	4548	4540
lk.catch.total	**402**	404	404	403	403	402
lk.discard.total	**648**	648	648	648	648	648
lk.index.fit.total	**168**	202	202	188	179	173
lk.catch.age.comp	**878**	887	887	883	882	880
lk.discards.age.comp	**539**	540	540	540	540	540
lk.survey.age.comp	**1475**	1475	1475	1482	1483	1481
lk.Recruit.devs	**420**	405	405	409	412	416
R0	**26431**	21165	21165	22381	23597	24975
R1970	**28663**	20774	20774	22374	24145	26267
mean_R	**21358**	14866	14866	16294	17798	19519
SSB0	**273763**	219221	219221	231813	244409	258676
SSB.1970	**297288**	112713	112713	140392	175604	225427
CV.SSB.1970	**0.14**	0.12	0.13	0.14	0.14	0.14
SSB1970/SSB0	**1.09**	0.51	0.51	0.61	0.72	0.87
SSB2009	**196339**	95340	95340	118945	143432	169545
CV.SSB2009	**0.14**	0.18	0.18	0.18	0.16	0.15
SSB2009/SSB0	**0.72**	0.43	0.43	0.51	0.59	0.66
F1970 (ave. 5-7)	**0.11**	0.18	0.18	0.16	0.14	0.12
CV.F1970(ave. 5-7)	**0.13**	0.12	0.12	0.13	0.13	0.13
F2009 (ave 5-7)	**0.07**	0.11	0.11	0.10	0.09	0.08
CV.F2009 (ave 5-7)	**0.16**	0.17	0.18	0.18	0.17	0.16
steepness	**0.66**	0.68	0.68	0.67	0.66	0.66
CV.steepness	**0.24**	0.12	0.13	0.16	0.18	0.21
Spring index q	**2.53E-05**	4.34E-05	4.34E-05	3.66E-05	3.19E-05	2.81E-05
Fall index q	**1.36E-05**	2.19E-05	2.19E-05	1.89E-05	1.67E-05	1.49E-05

Table 14 (cont.).

Model estimate	Index.sel(9+)=0.3	Est Index.sel(9+)	Index.sel(9+)=0.2	Index.sel(9+)=0.1	Index.sel(6-9+)=1 ("Flat")	base, M=0.15	Base including CAN 5Z landings
lk.total	4521	4515	4516	4525	4567	4540	4523
lk.catch.total	401	401	401	401	405	403	408
lk.discard.total	648	648	648	648	648	648	648
lk.index.fit.total	165	164	165	168	216	184	168
lk.catch.age.comp	879	877	877	878	889	886	880
lk.discards.age.comp	541	538	538	537	540	540	533
lk.survey.age.comp	1458	1454	1452	1455	1466	1483	1466
lk.Recruit.devs	428	432	433	437	402	396	419
R0	29810	31580	32109	34235	20327	16844	26552
R1970	34761	37927	39000	42225	19606	15957	28589
mean_R	25649	27904	28624	31079	13838	12046	21316
SSB0	308762	327085	332574	354585	210533	296643	275016
SSB.1970	630853	928990	1159244	3044910	94254	159427	285724
CV.SSB.1970	0.15	0.12	0.15	0.15	0.12	0.13	0.15
SSB1970/SSB0	2.04	2.84	3.49	8.59	0.45	0.54	1.04
SSB2009	255240	287344	296970	331614	76731	134298	177337
CV.SSB2009	0.14	0.18	0.14	0.14	0.18	0.17	0.16
SSB2009/SSB0	0.83	0.88	0.89	0.94	0.36	0.45	0.64
F1970 (ave. 5-7)	0.08	0.07	0.07	0.06	0.20	0.18	0.11
CV.F1970(ave. 5-7)	0.15	0.12	0.15	0.16	0.12	0.12	0.14
F2009 (ave 5-7)	0.06	0.05	0.05	0.05	0.13	0.10	0.08
CV.F2009 (ave 5-7)	0.16	0.17	0.17	0.17	0.17	0.17	0.17
steepness	0.68	0.73	0.75	1.00	0.70	0.70	0.70
CV.steepness	0.31	0.12	0.38	0.04	0.12	0.15	0.23
Spring index q	2.12E-05	1.90E-05	1.91E-05	1.77E-05	5.05E-05	4.17E-05	2.68E-05
Fall index q	1.12E-05	1.03E-05	1.00E-05	9.23E-06	2.46E-05	2.15E-05	1.38E-05

Table C15. Total commercial landings from the base model (column 1) and Canadian landings in area 5Zc. The total landings in column 3 were used in a sensitivity analysis.

Year	Total Commercial Landings (mt) in US areas 5 and 6	Canadian landings (mt) in area 5Zc	Total landings	(5Zc landings)/ Total landings
1970	11555	0	11555	0
1971	14319	0	14319	0
1972	12995	0	12995	0
1973	13080	0	13080	0
1974	12014	0	12014	0
1975	13482	0	13482	0
1976	12731	0	12731	0
1977	15516	0	15516	0
1978	21512	0	21512	0
1979	17645	0	17645	0
1980	22201	0	22201	0
1981	21816	0	21816	0
1982	19335	4430	23765	0.19
1983	18225	3301	21526	0.15
1984	20947	1199	22146	0.05
1985	20956	911	21867	0.04
1986	24994	1538	26532	0.06
1987	20251	2096	22347	0.09
1988	14830	2403	17233	0.14
1989	10553	1385	11938	0.12
1990	9559	1740	11299	0.15
1991	7886	1715	9601	0.18
1992	7184	3036	10220	0.30
1993	5674	4193	9867	0.42
1994	3763	3327	7090	0.47
1995	3352	1004	4356	0.23
1996	2962	1200	4162	0.29
1997	4264	1231	5495	0.22
1998	5572	1857	7429	0.25
1999	4590	996	5586	0.18
2000	4043	1197	5240	0.23
2001	4109	1569	5678	0.28
2002	3580	1616	5196	0.31
2003	4794	1347	6141	0.22
2004	5070	2047	7117	0.29
2005	6509	1740	8249	0.21
2006	6067	848	6915	0.12
2007	8372	552	8924	0.06
2008	9965	389	10354	0.04
2009	7477	280	7757	0.04

Table C16. Model results (kmt) for the ASAP base pollock model and three SCAA sensitivity models, showing the point estimates, and medians and 90% PIs. SCAA1 downweights the CAA proportions data whereas SCAA2 gives these data full weight. SCAA3 duplicates SCAA2 but fixes the 9+ survey selectivity at its estimated value when computing posterior distributions. The SSB_{MSY} and MSY results are F40%-based proxies. Further detail is given in the text.

	ASAP			SCAA1			SCAA2			SCAA3		
	est.	med.	90% PI	est.	med.	90% PI	est.	med.	90% PI	est.	med.	90% PI
SSB0	273	253	(232; 329)	395	968	(388; 2806)	446	474	(343; 794)	446	479	(359; 779)
SSB1970	297	289	(228; 360)	244	645	(206; 2313)	365	340	(208; 660)	365	383	(267; 687)
SSB2009	196	193	(153; 246)	233	624	(204; 2113)	328	325	(209; 613)	328	355	(249; 640)
SSBMSY	-	91	(71; 118)	100	97	(35; 356)	112	85	(60; 140)	112	116	(87; 188)
MSY	-	16.2	(11.8; 23.2)	16.4	13.5	(4.7; 49.4)	18.1	13.8	(9.6; 22.5)	18.1	18.6	(14.0; 30.0)

Table C17a. Percentiles of Pollock spawning stock biomass (000s mt) for projections at Fstatus quo, 0.75*F40%, and F40%.

	F-status-quo = 0.07 (average F on ages 5-7)								
YEAR	1%	5%	10%	25%	50%	75%	90%	95%	99%
2010	138.5	153.8	160.8	175.9	194.3	213.5	233.0	249.5	270.7
2011	130.7	143.5	149.5	163.2	179.8	196.6	215.6	229.8	250.1
2012	127.1	137.6	143.6	156.4	171.6	187.0	204.5	218.0	237.6
2013	123.6	133.9	140.5	152.5	166.6	181.4	198.0	209.4	228.6
2014	124.1	134.0	140.2	151.9	165.0	179.2	194.9	205.0	223.8
2015	125.5	135.2	141.4	152.4	164.9	178.8	193.7	202.8	221.3
2016	126.5	136.7	142.6	153.2	165.8	179.8	194.1	203.1	221.0
2017	126.5	136.8	142.7	153.3	166.2	180.5	194.9	204.1	221.8

	0.75*F40% = 0.19 (average F on ages 5-7)								
YEAR	1%	5%	10%	25%	50%	75%	90%	95%	99%
2010	138.5	153.8	160.8	175.9	194.3	213.5	233.0	249.5	270.7
2011	122.5	134.2	139.9	152.8	168.3	184.3	202.2	214.8	234.0
2012	112.3	121.1	126.6	138.0	151.2	165.1	180.7	191.7	209.8
2013	104.1	112.8	118.1	128.5	140.0	152.6	166.5	176.2	192.7
2014	100.1	108.0	113.0	122.4	132.8	144.3	156.8	165.0	180.8
2015	96.9	104.7	109.3	117.8	127.6	138.5	149.8	157.1	171.4
2016	93.7	101.4	105.8	113.9	123.5	134.4	145.5	152.6	166.1
2017	90.2	97.8	102.2	110.1	120.0	131.2	142.5	149.7	163.6

	F40% = 0.25 (average F on ages 5-7)								
YEAR	1%	5%	10%	25%	50%	75%	90%	95%	99%
2010	138.5	153.8	160.8	175.9	194.3	213.5	233.0	249.5	270.7
2011	118.5	129.6	135.2	147.7	162.6	178.0	195.5	207.6	226.2
2012	105.3	113.4	118.9	129.7	142.0	155.0	169.6	180.0	197.1
2013	95.7	103.4	108.4	117.9	128.5	140.0	152.8	161.4	177.0
2014	90.0	97.1	101.7	110.0	119.4	129.8	141.0	148.4	162.8
2015	85.4	92.4	96.5	103.9	112.6	122.4	132.4	138.9	151.5
2016	81.0	87.7	91.6	98.6	107.3	117.0	127.0	133.5	145.7
2017	76.6	83.2	86.9	93.9	102.8	112.8	123.2	129.7	142.4

Table C17b. Percentiles of catch (000s mt) for projections at Fstatus quo, 0.75*F40%, and F40%.

	F-status-quo = 0.07 (average F on ages 5-7)								
YEAR	1%	5%	10%	25%	50%	75%	90%	95%	99%
2010	5.8	6.4	6.7	7.3	8.1	8.7	9.6	10.2	11.2
2011	5.5	6.0	6.2	6.8	7.5	8.1	8.8	9.4	10.4
2012	5.3	5.7	6.0	6.6	7.2	7.8	8.5	9.0	9.8
2013	5.5	6.1	6.3	6.9	7.5	8.2	9.0	9.4	10.3
2014	5.9	6.5	6.8	7.3	8.0	8.8	9.6	10.1	11.1
2015	6.3	6.8	7.1	7.7	8.4	9.2	10.0	10.5	11.6
2016	6.4	7.0	7.3	7.8	8.5	9.3	10.2	10.7	11.7
2017	6.1	6.6	7.0	7.6	8.4	9.3	10.5	11.3	12.6

	0.75*F40% = 0.19 (average F on ages 5-7)								
YEAR	1%	5%	10%	25%	50%	75%	90%	95%	99%
2010	14.3	15.8	16.5	17.9	19.8	21.5	23.6	25.0	27.6
2011	12.4	13.5	14.1	15.3	16.9	18.4	20.0	21.2	23.4
2012	11.4	12.3	12.9	14.1	15.4	16.8	18.3	19.4	21.0
2013	11.4	12.5	13.1	14.2	15.6	17.0	18.5	19.5	21.3
2014	11.8	12.9	13.5	14.6	16.0	17.6	19.2	20.2	22.3
2015	12.2	13.3	13.9	15.0	16.3	17.9	19.4	20.4	22.5
2016	12.1	13.1	13.7	14.8	16.1	17.7	19.4	20.6	22.7
2017	11.0	12.1	12.7	14.0	15.6	17.5	19.8	21.4	24.0

	F40% = 0.25 (average F on ages 5-7)								
YEAR	1%	5%	10%	25%	50%	75%	90%	95%	99%
2010	18.6	20.4	21.3	23.2	25.7	27.9	30.5	32.4	35.8
2011	15.3	16.7	17.5	19.0	21.0	22.8	24.8	26.3	29.0
2012	13.8	14.9	15.6	17.1	18.6	20.3	22.2	23.4	25.4
2013	13.5	14.9	15.5	16.9	18.4	20.1	22.0	23.1	25.3
2014	13.7	15.0	15.7	17.0	18.6	20.5	22.4	23.5	26.0
2015	14.1	15.3	16.0	17.2	18.7	20.6	22.3	23.5	25.9
2016	13.7	14.9	15.6	16.8	18.3	20.2	22.2	23.6	26.2
2017	12.3	13.5	14.2	15.7	17.5	19.8	22.6	24.4	27.4

Figures

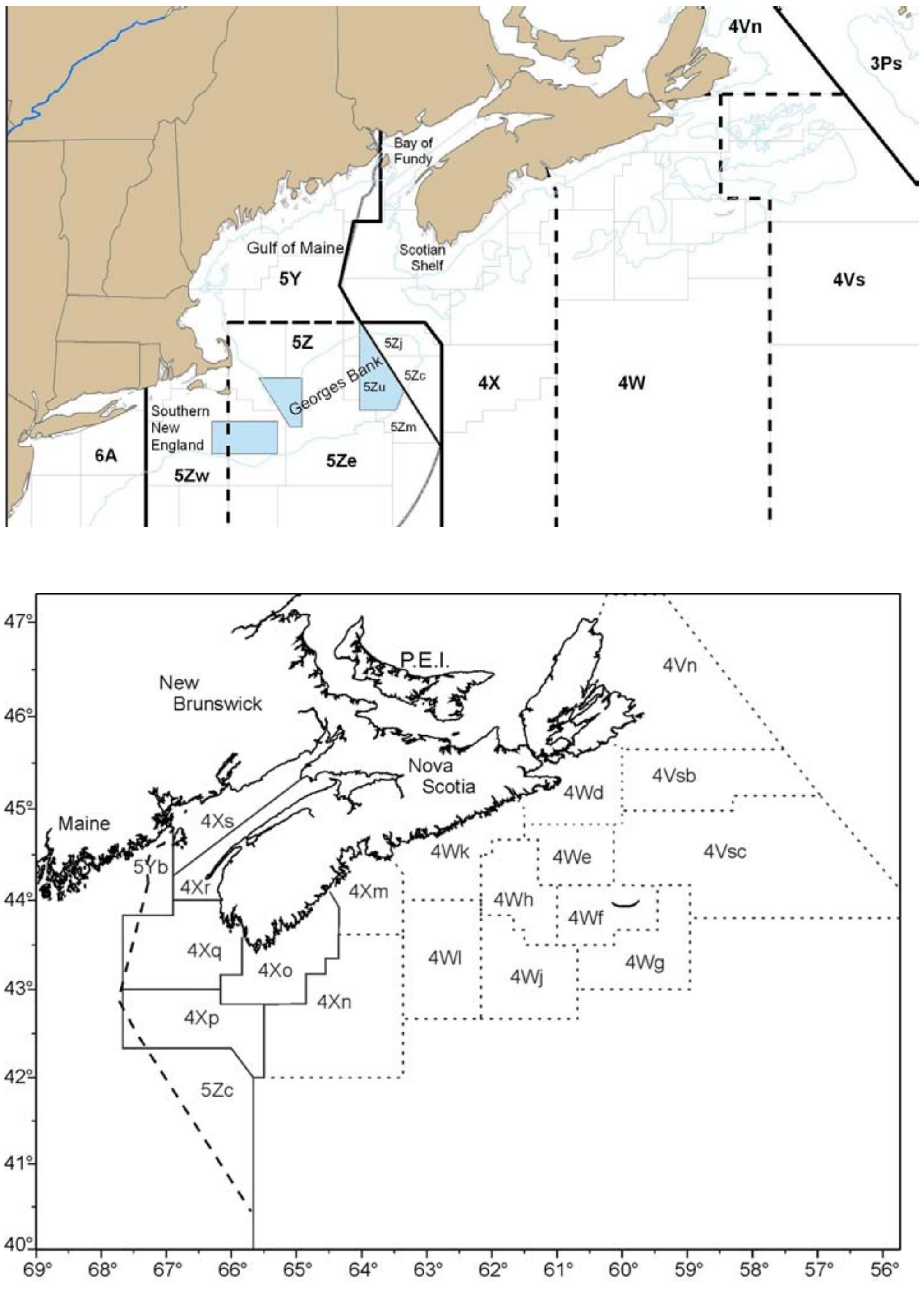


Figure C1. NAFO areas.

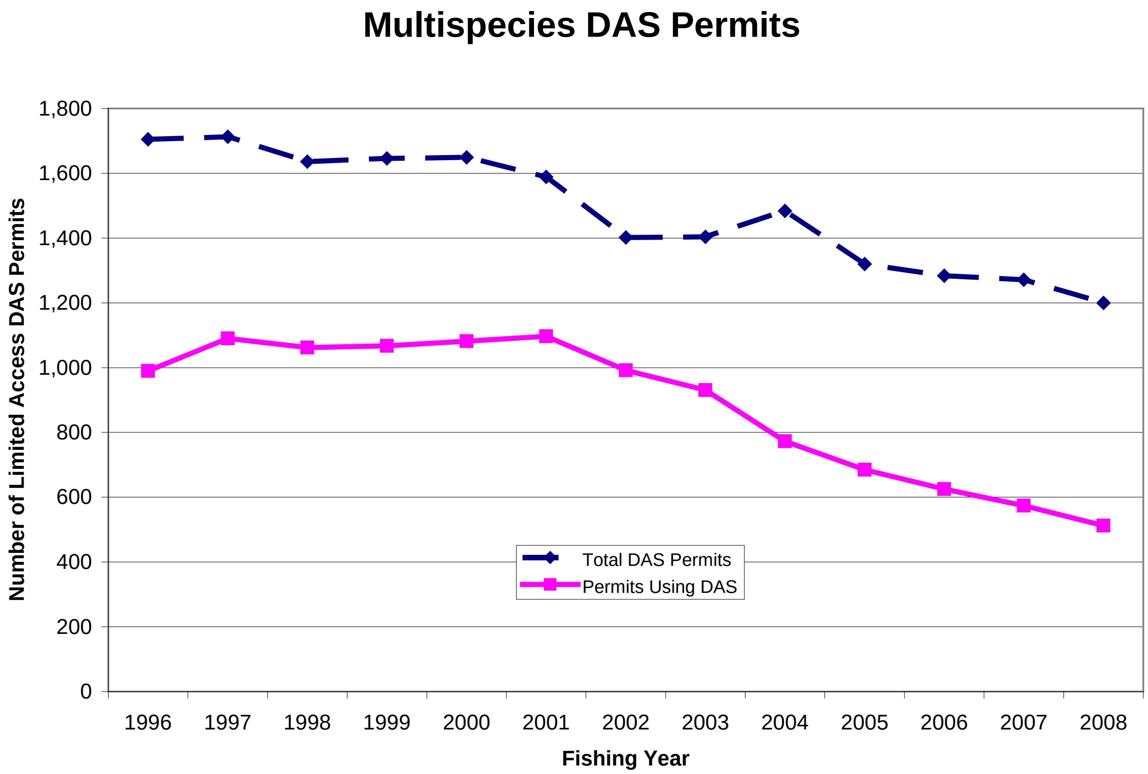


Figure C2. Multispecies DAS permits issued and permits using DAS, 1996 – 2008.

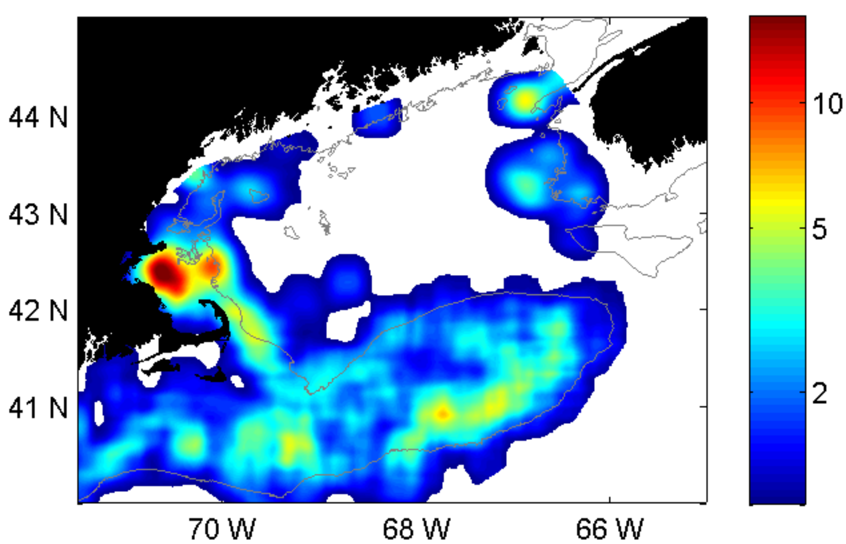
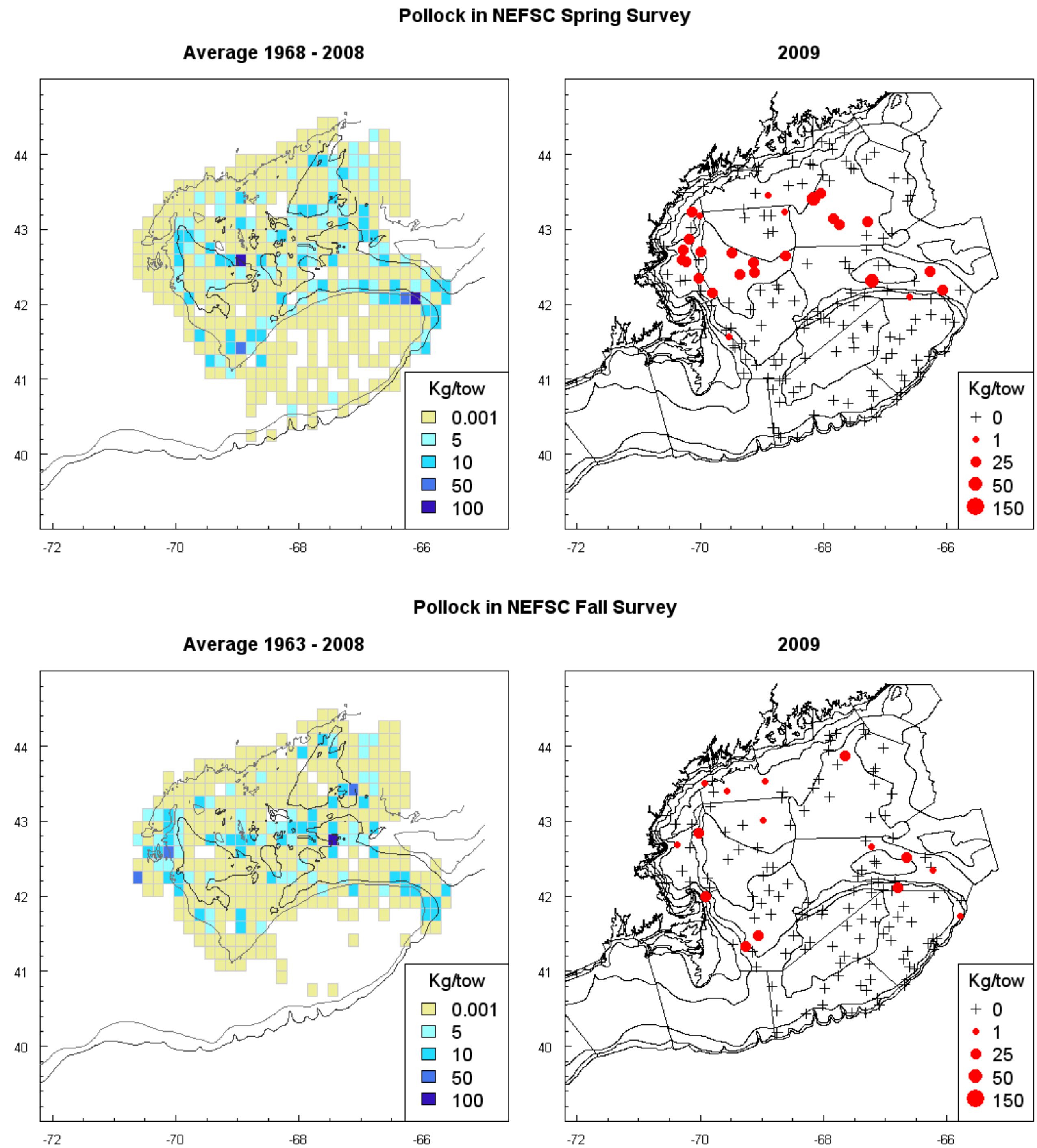
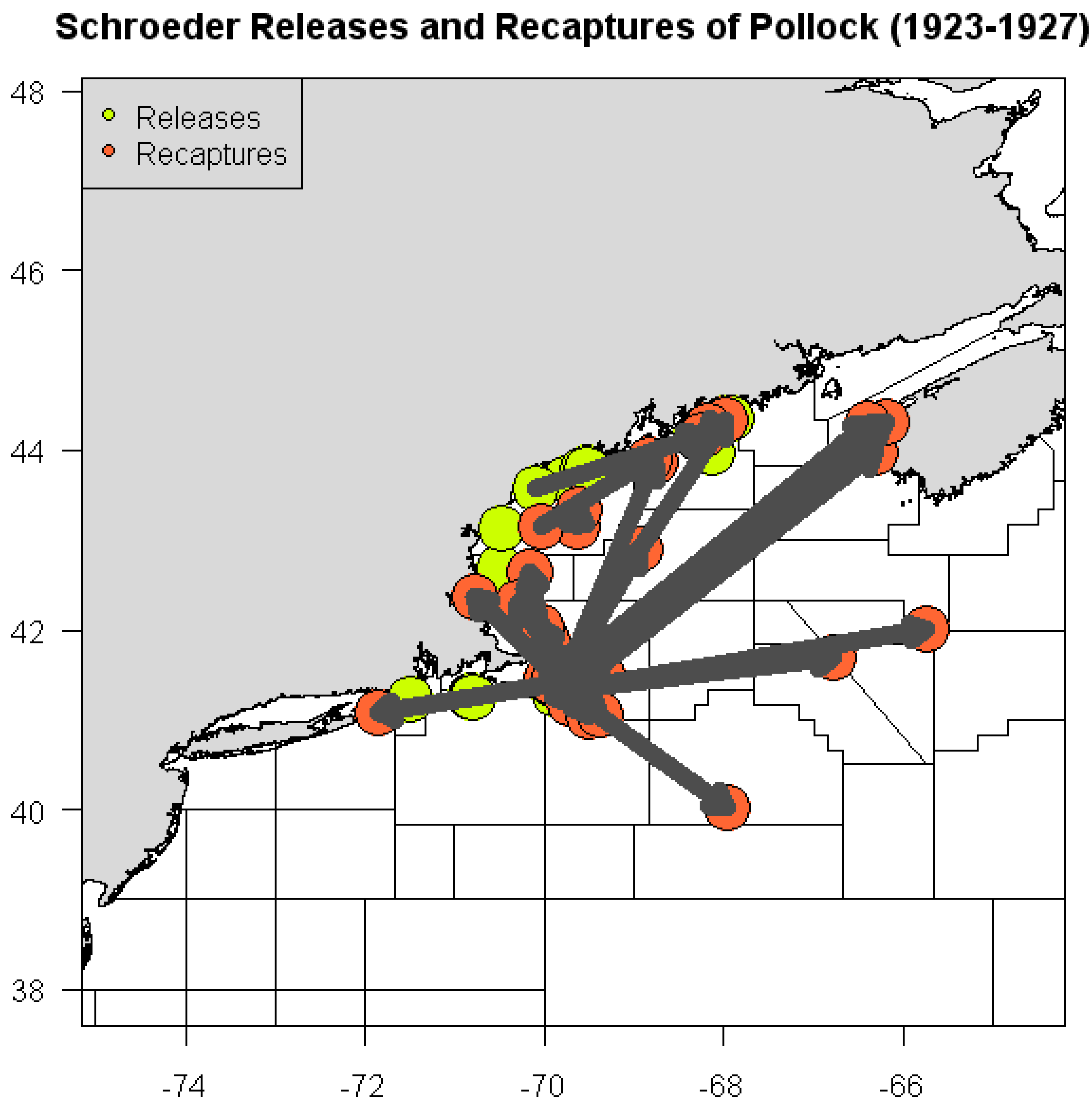


Figure C3. Spatial distribution of pollock larvae from January – March (1978-present).



C4. NEFSC bottom trawl survey distributions for spring (top) and fall (bottom) and the most recent survey (2009, right panels).



C5. Preliminary analysis of schroeder tag releases and recaptures. The scale of the release and recapture circles is large, as are the connecting arrows, to convey the lack of fine-scale resolution on those locations.

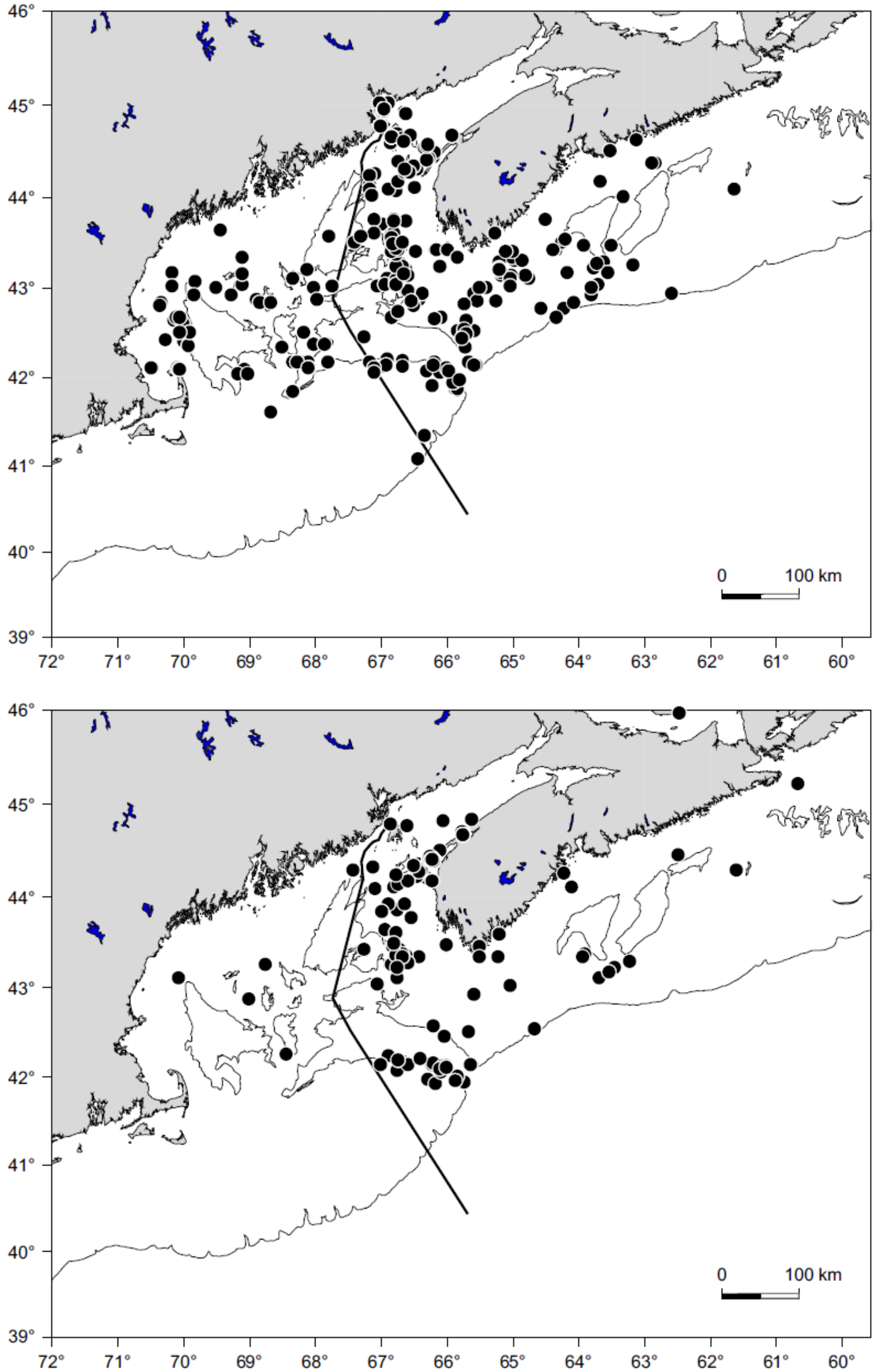


Figure C6a. The location of recaptures of marked pollock released in the eastern side of the Bay of Fundy (statistical Unit Area 4Xr, top panel), and the location of recaptured marked pollock released in the western side of the Bay of Fundy (statistical Unit Area 4Xs, bottom panel). (Figure 10 from Neilson et al. 2006; reprinted with permission from J.D.Neilson).

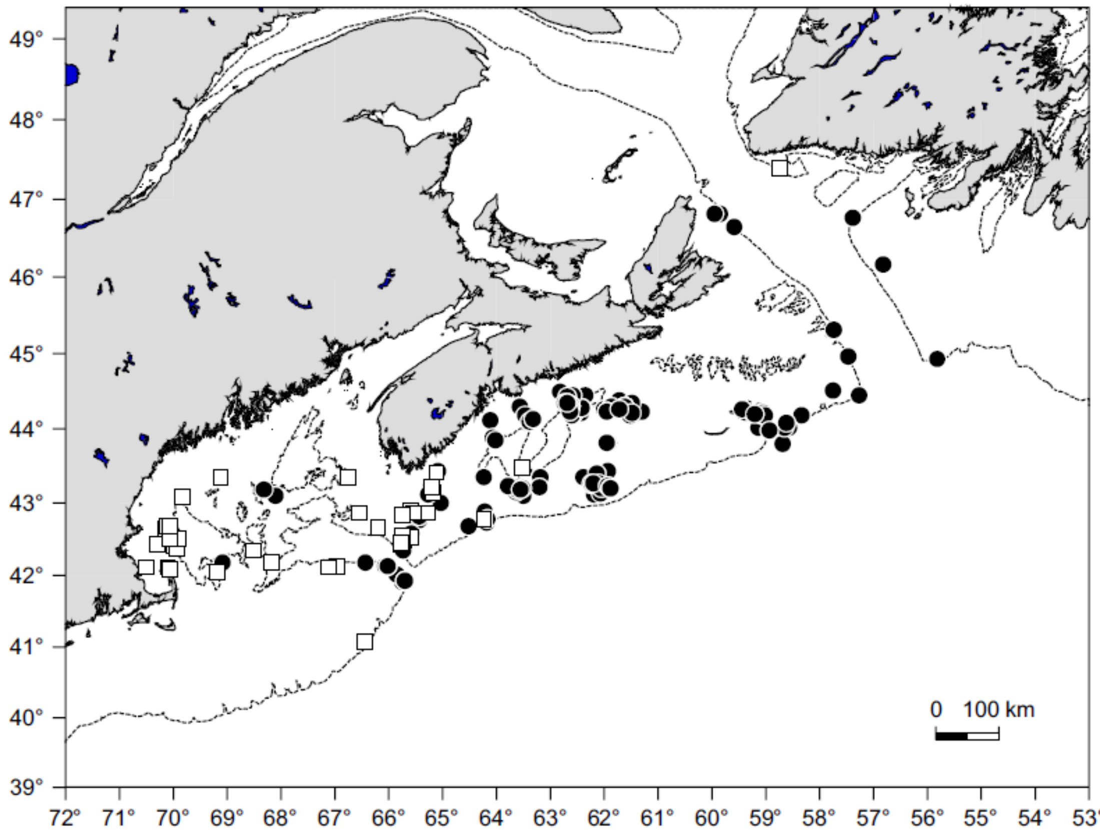
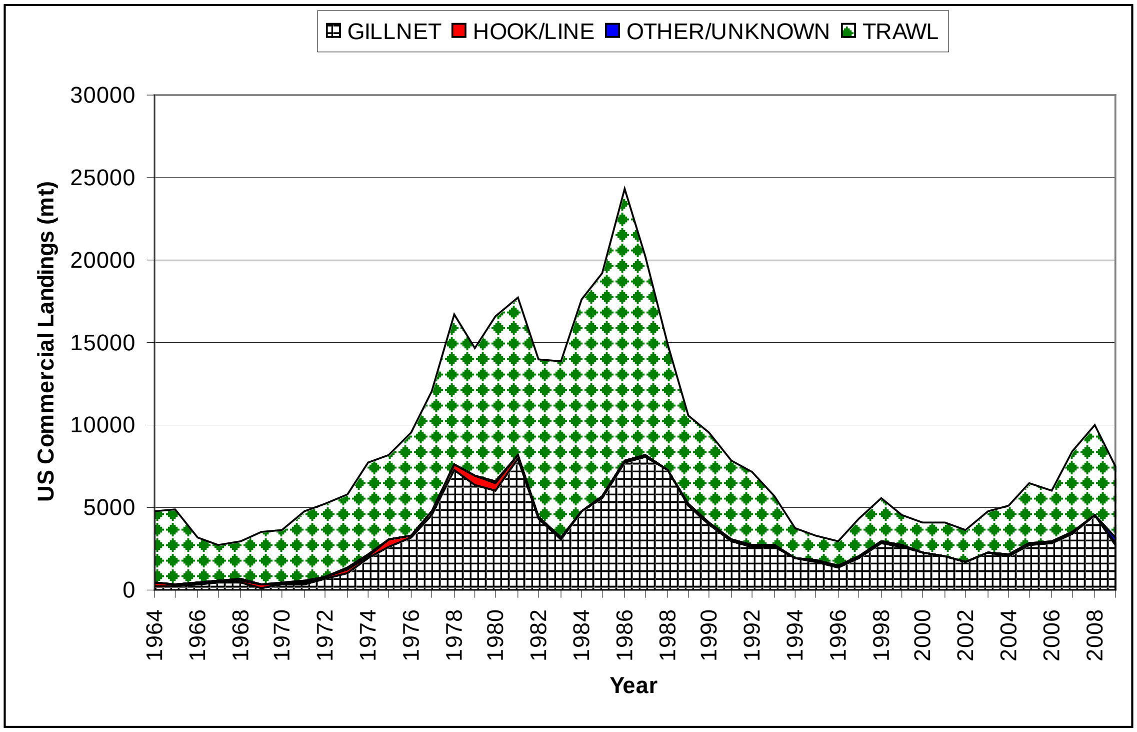
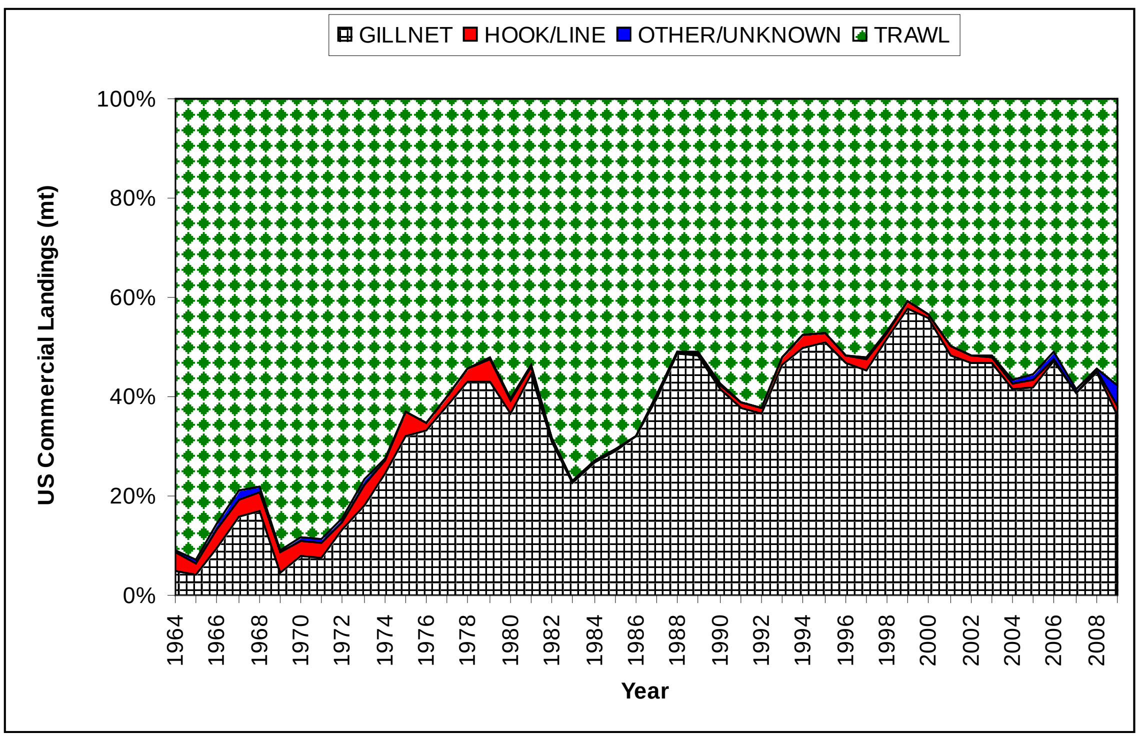
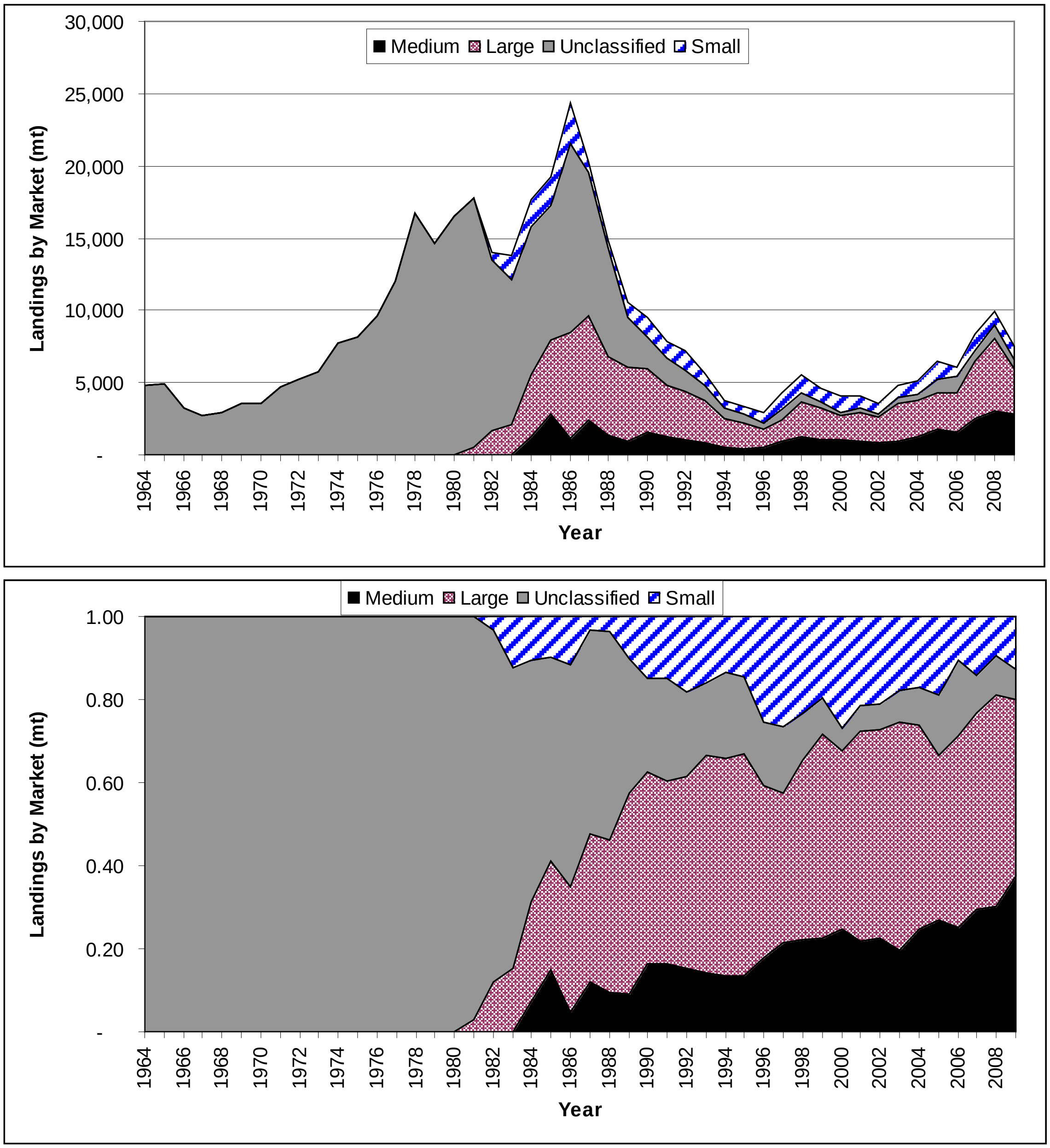


Figure C6b. Locations of recaptures of presumed spawners (>50 cm; recaptures made from November to February). Locations marked by an open square signify fish that were released near the western extremity of the management unit (4Xs; see Figure C1), and those locations marked with a filled circle signify fish that were released near the eastern extremity of the management unit (4Wd). (Figure 12 from Neilson et al. 2006; reprinted with permission from J.D.Neilson).

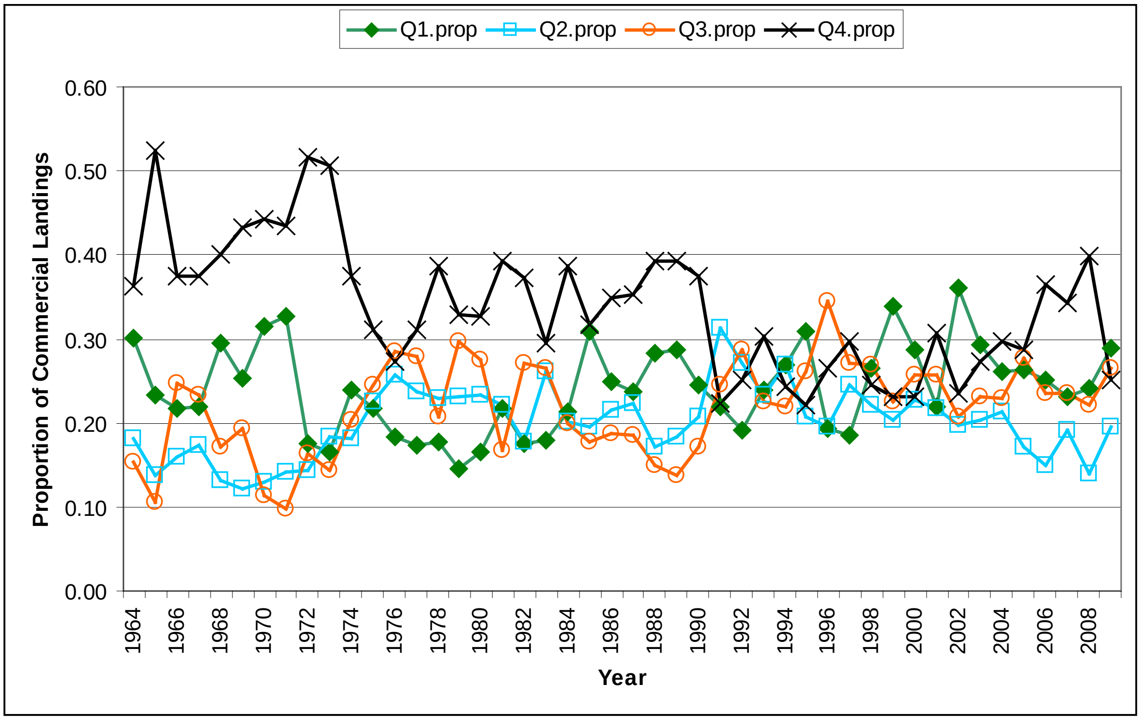




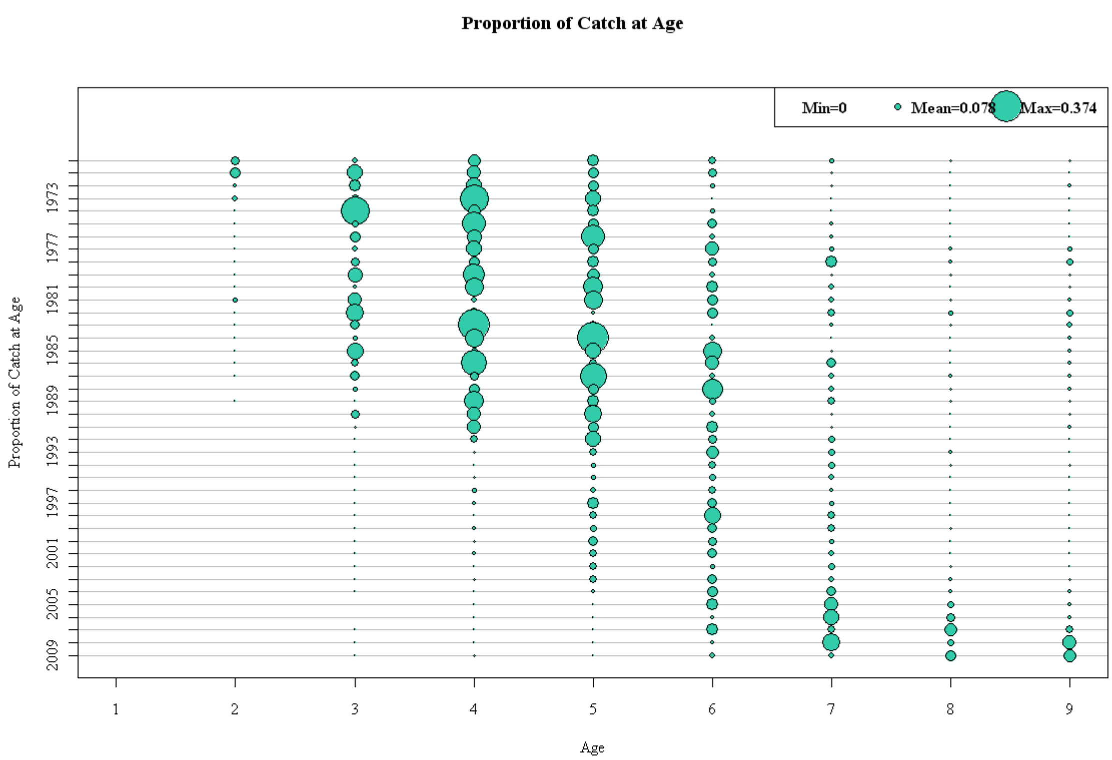
C7. US Commercial landings of pollock (mt) by gear.



C8. US commercial landings of pollock (mt) by market category.



C9. US commercial landings of pollock by quarter.



C10. Total commercial landings at age of pollock expressed as a proportion of total annual landings.

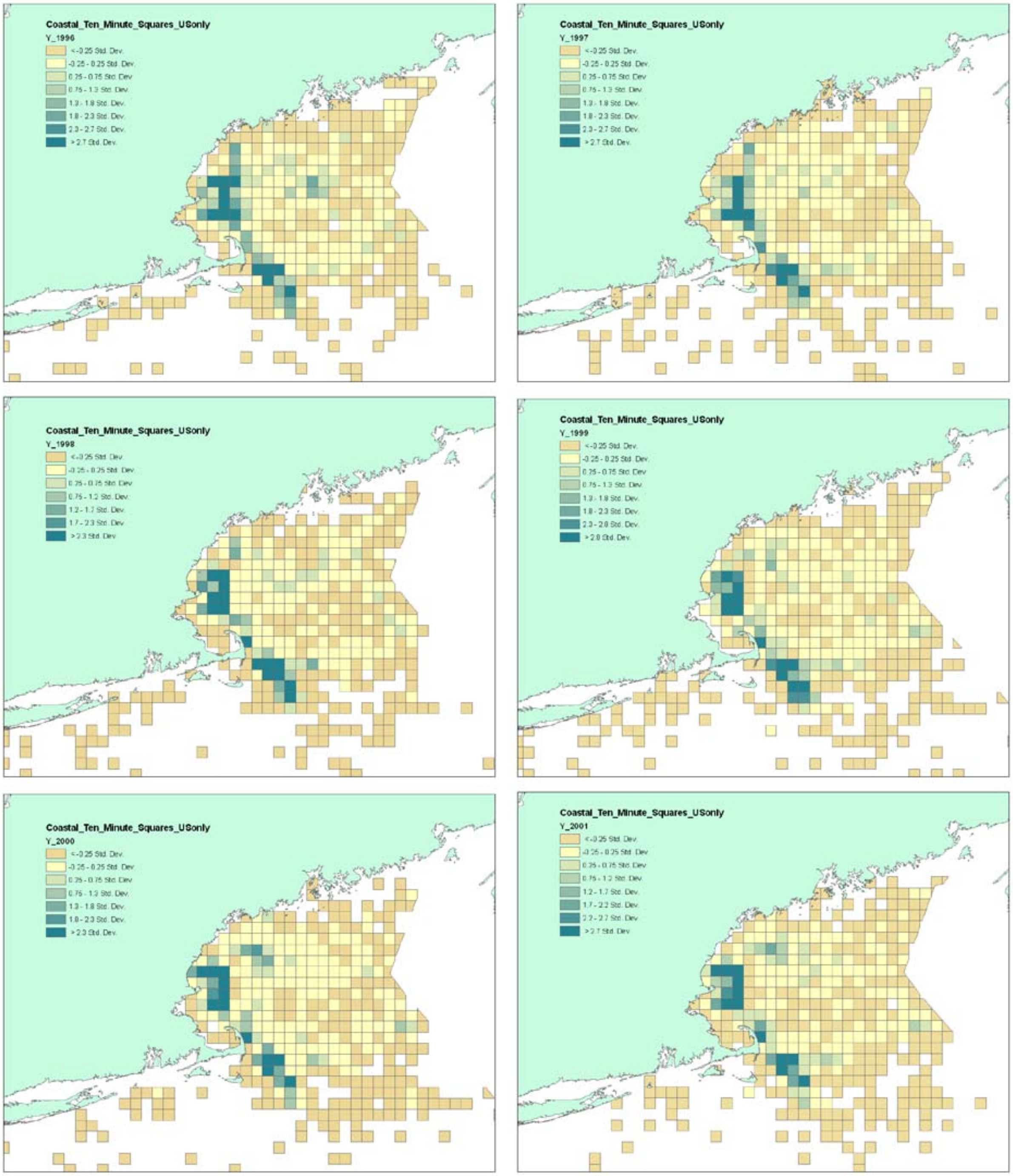


Figure C11. Sum of Trips Landing Pollock by VTR Area, 1996-2008 (Standard Deviation)

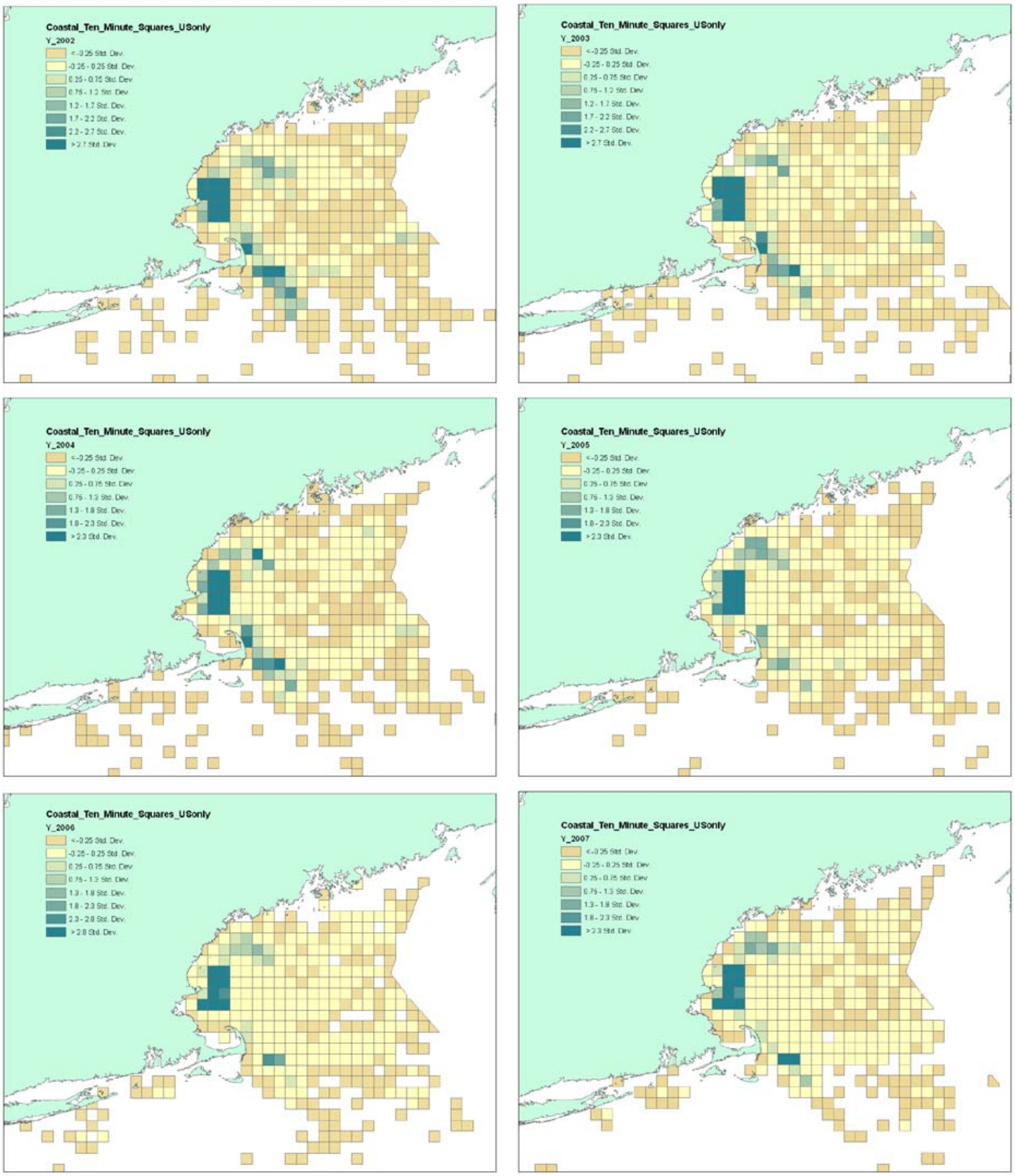


Figure C11. (cont.)

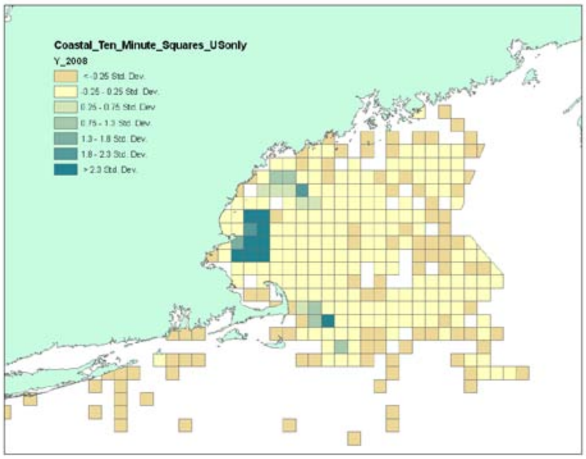


Figure C11. (cont)

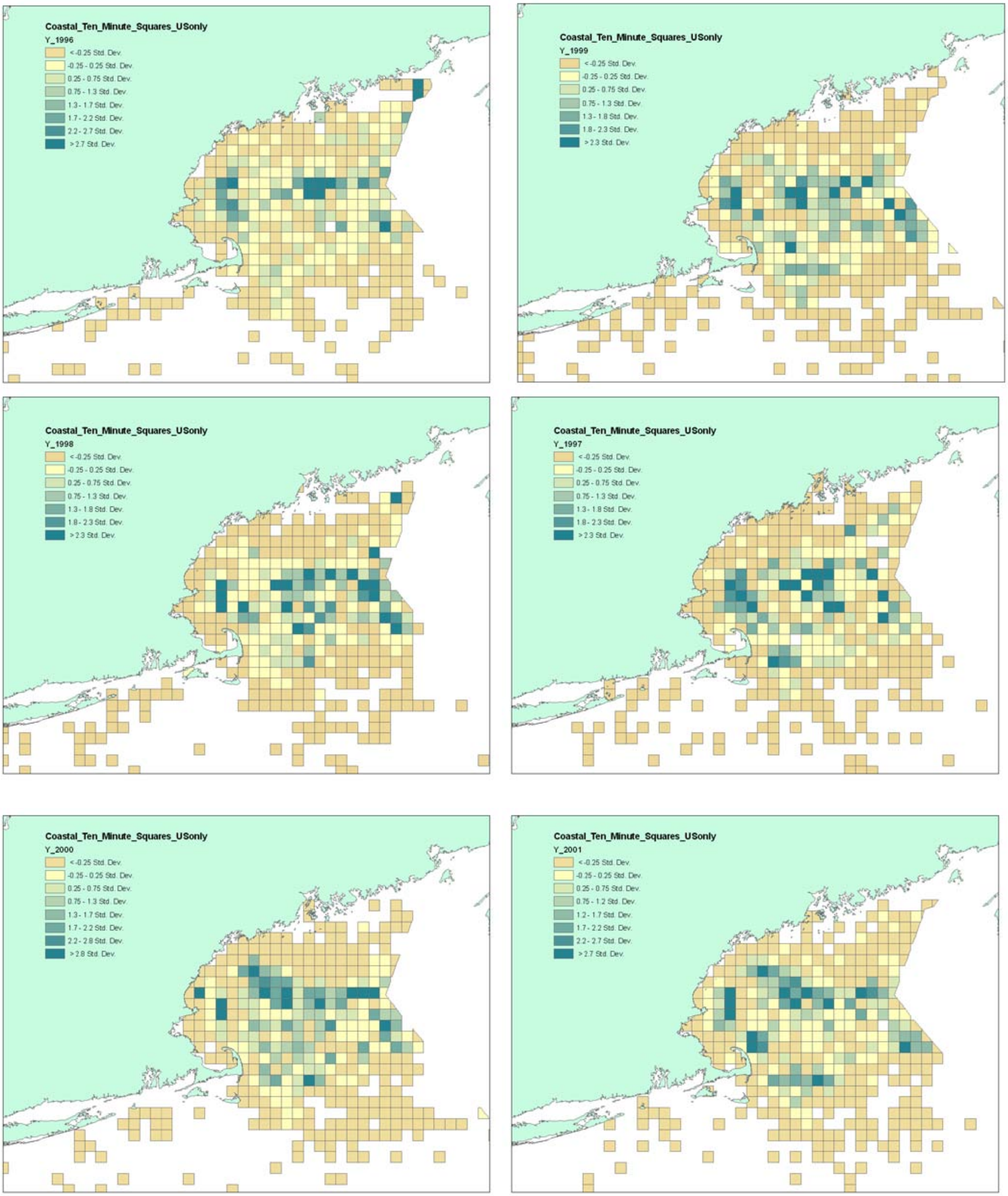


Figure C12. Pollock landed by VTR area, 1996-2008 (Standard Deviation).

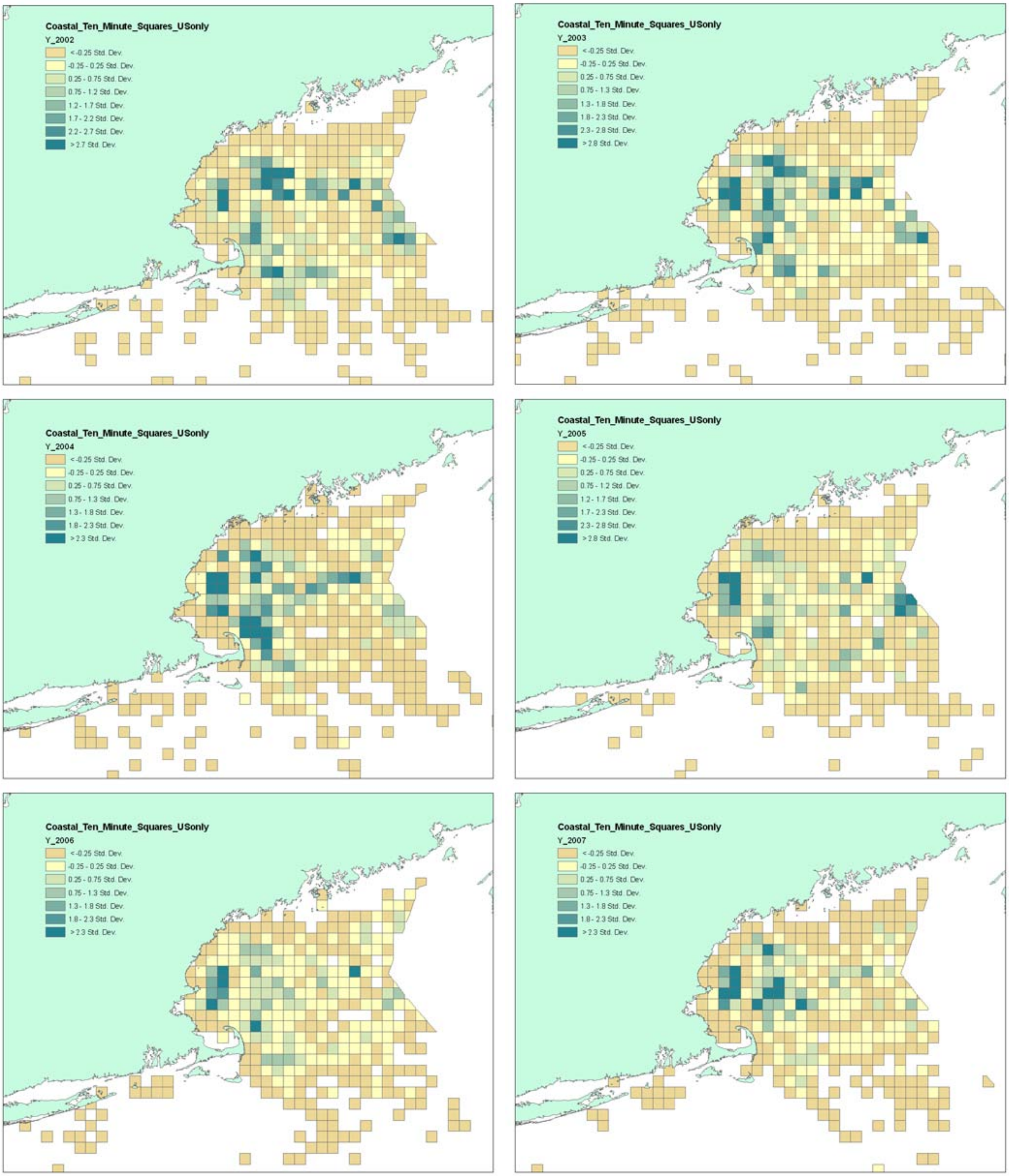


Figure C12. (cont)

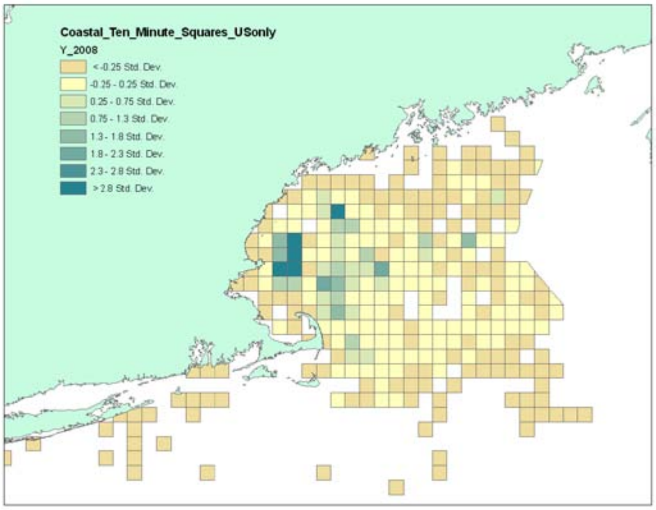


Figure C12. (cont.)

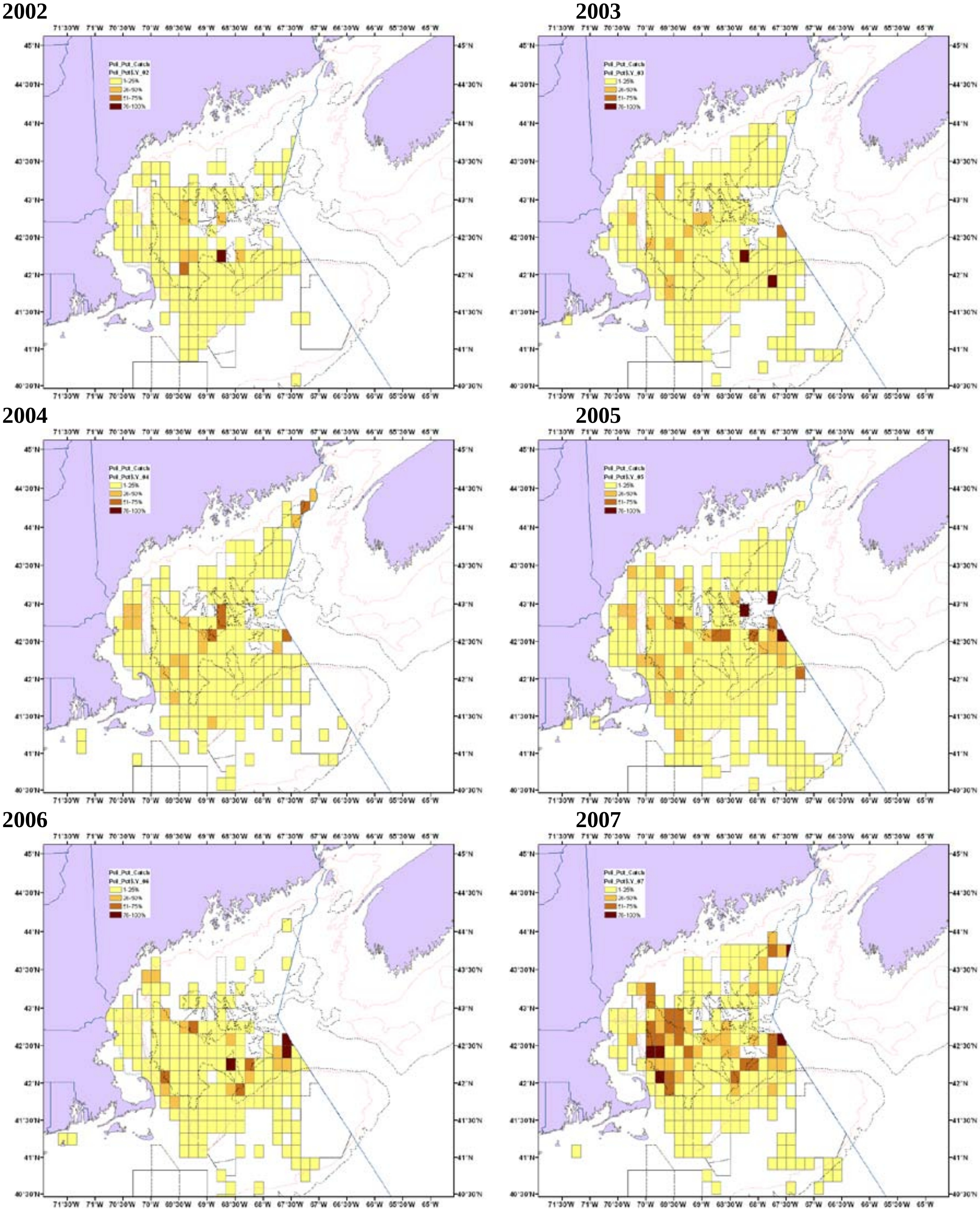


Figure C13. Pollock as a percent of the observed trawl catch in a ten-minute square, 2002-2009.

2008 **2009**

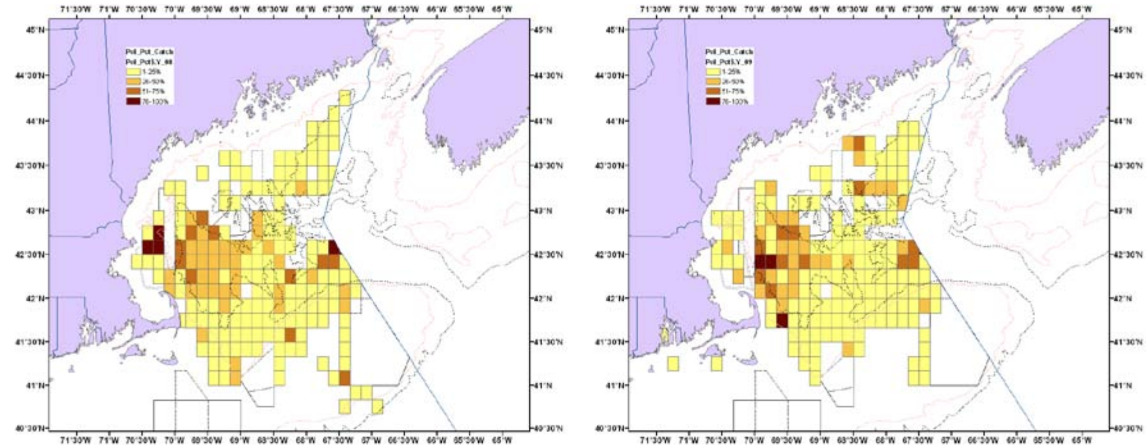


Figure C13. (cont.)

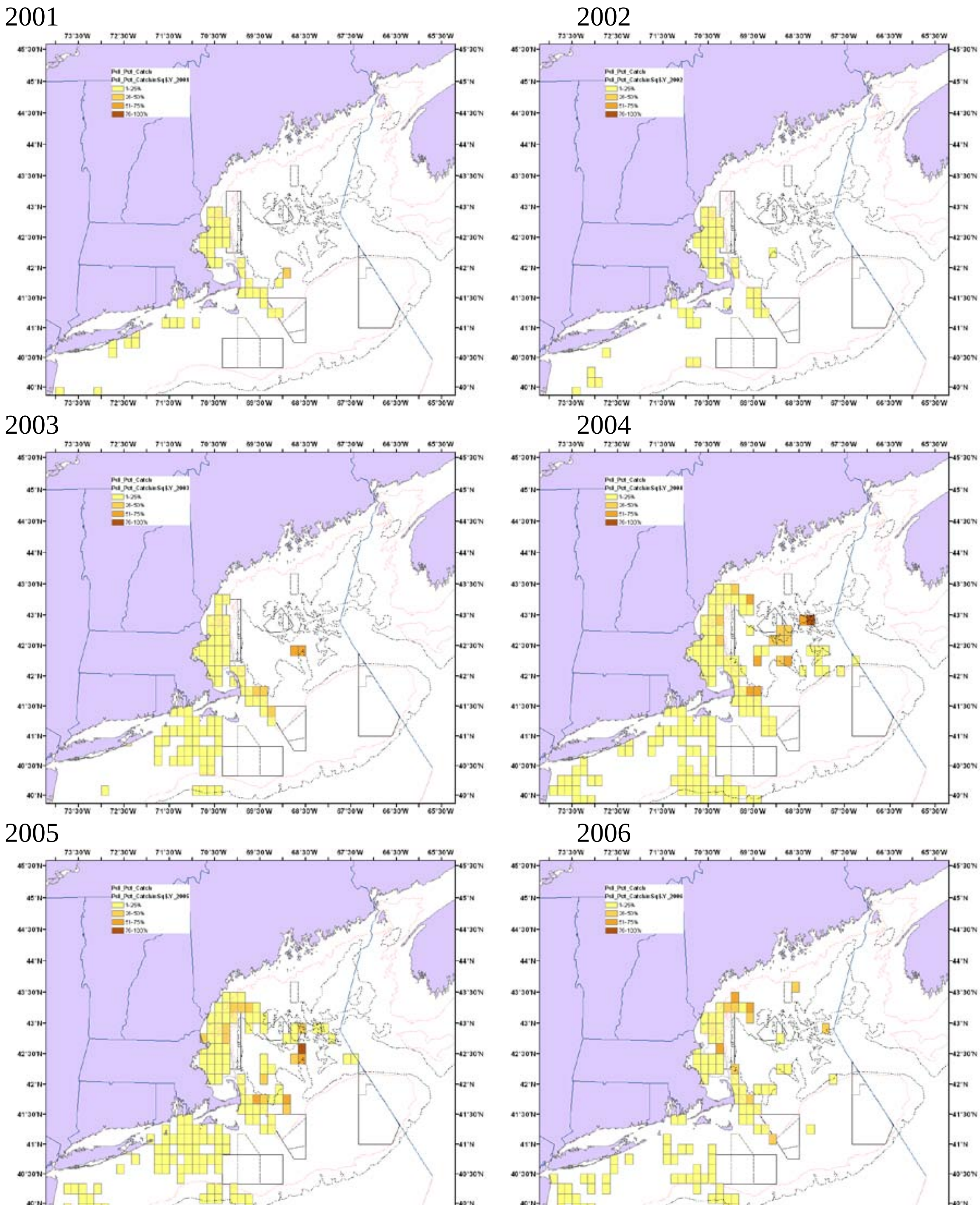
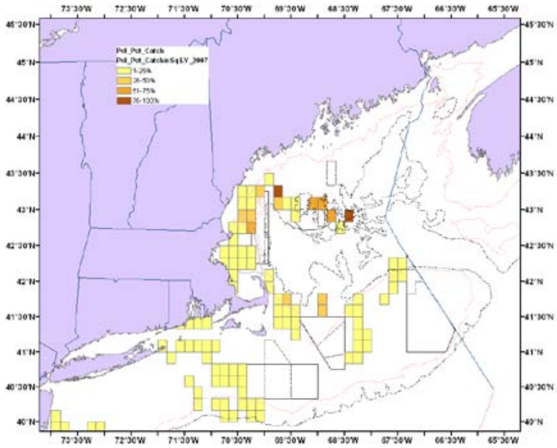
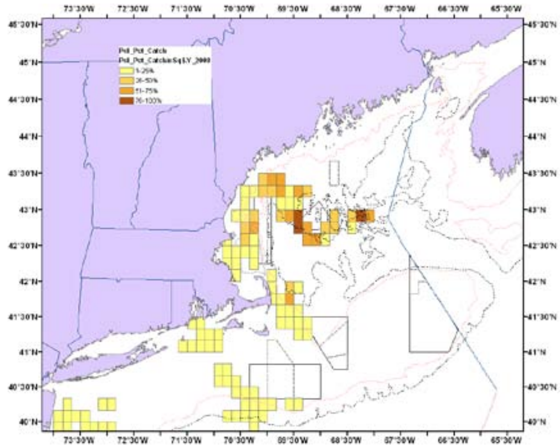


Figure C14. Pollock as percent of sink gillnet catch, 2001 – 2009.

2007



2008



2009

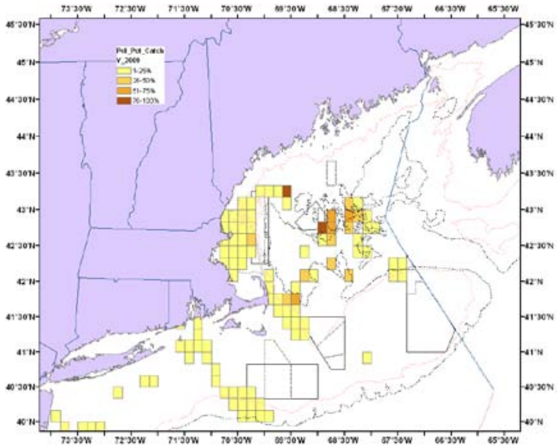
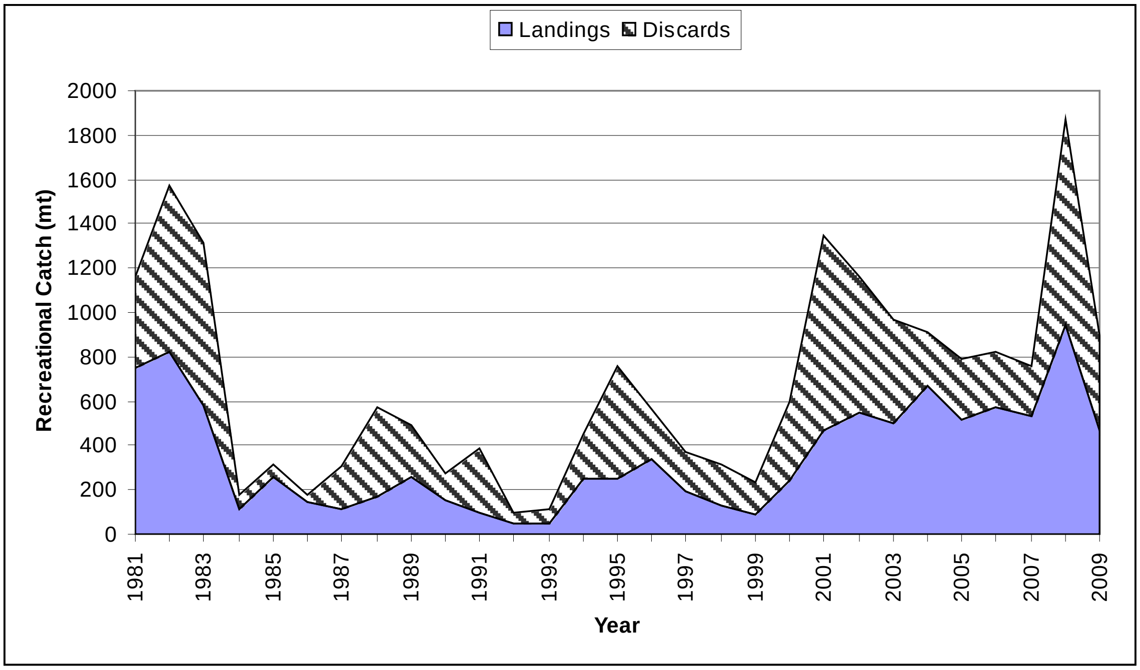
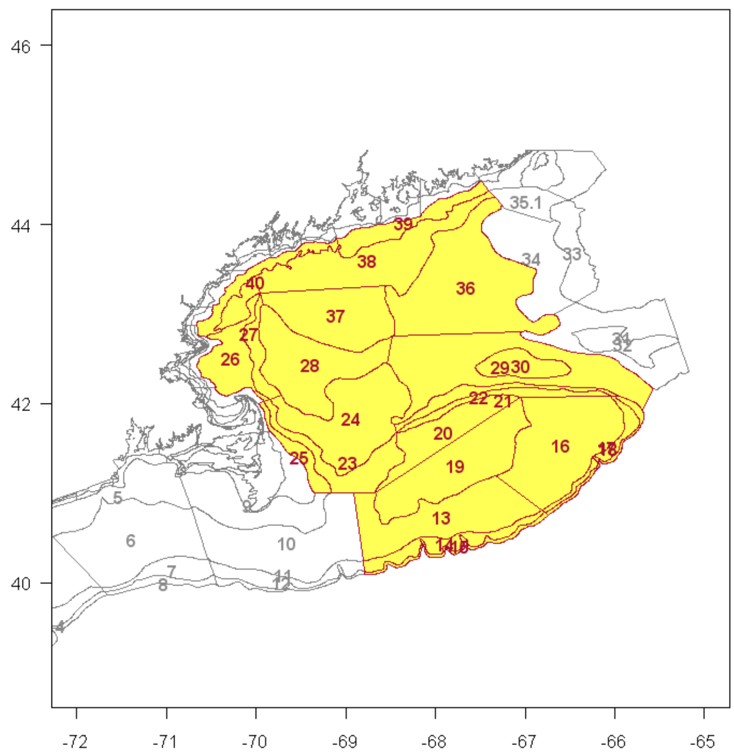


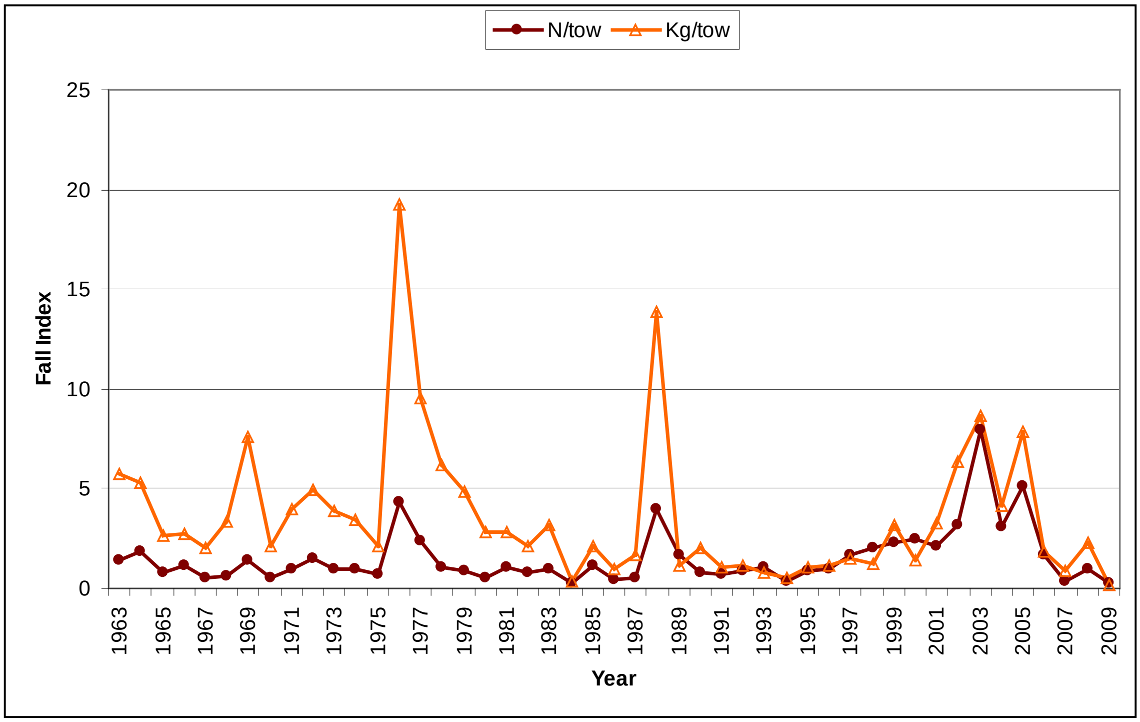
Figure C14. (cont.)



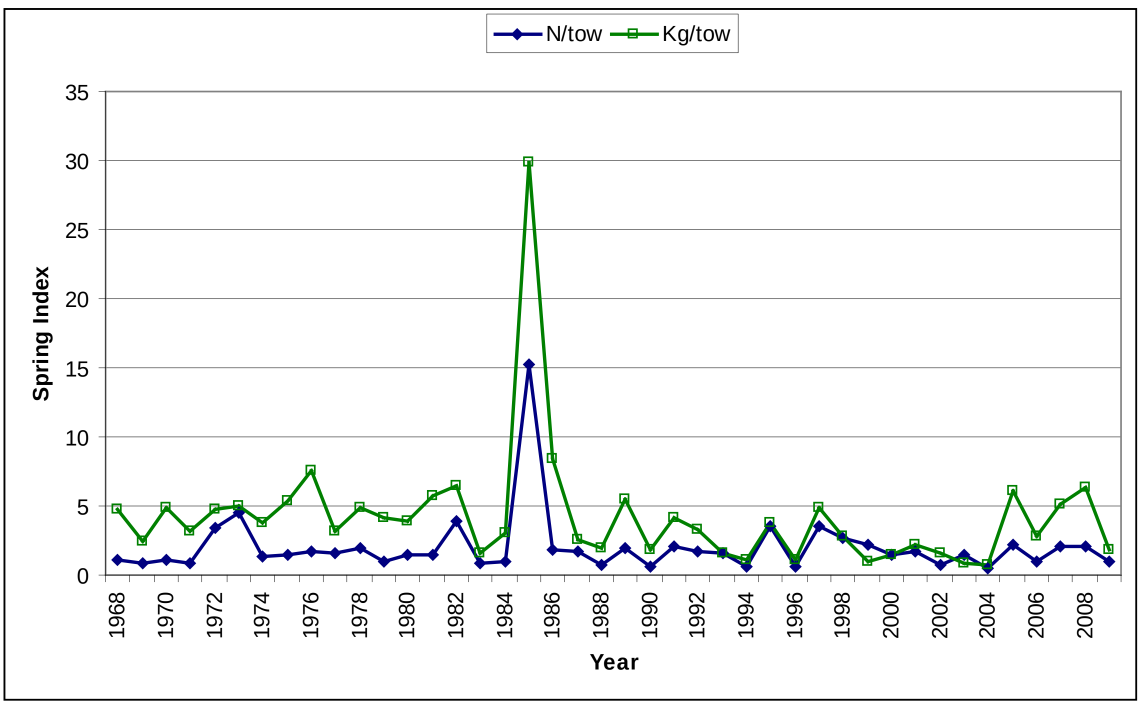
C15. US recreational catch (mt) of pollock.



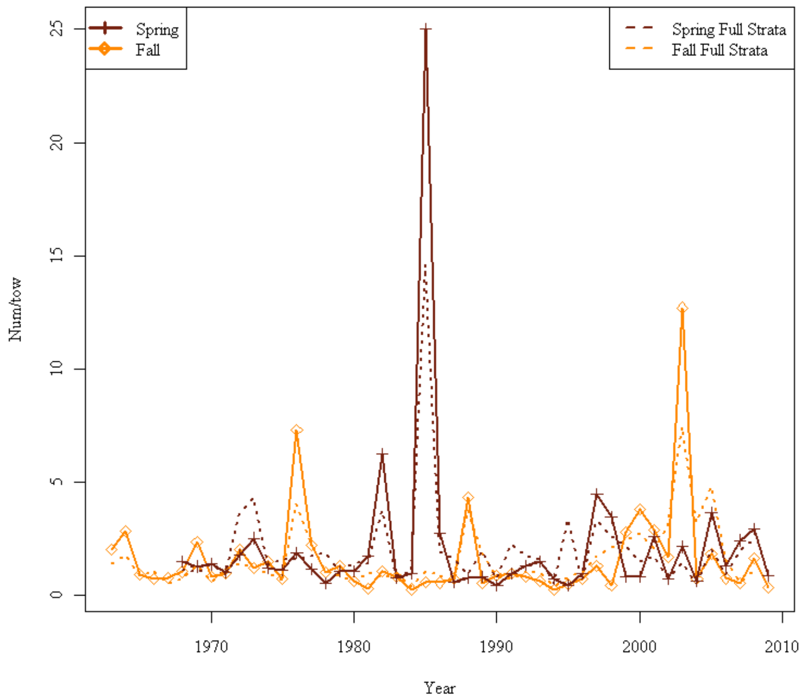
C16. NEFSC bottom trawl survey strata used to represent the pollock stock.

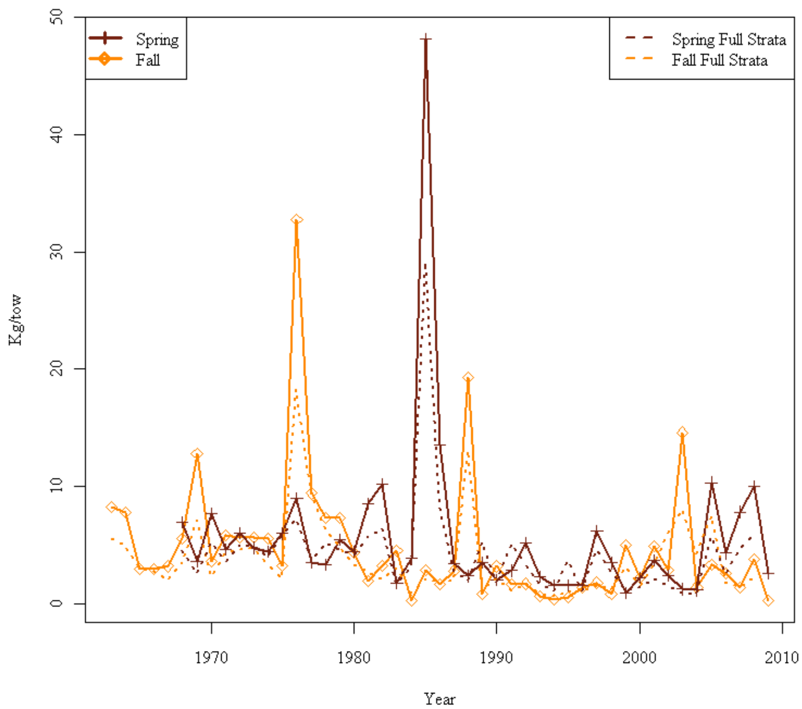


C17. NEFSC bottom trawl fall survey index.



C18. NEFSC bottom trawl spring survey index.





C19. Comparison of NEFSC spring and fall bottom trawl survey indices for Pollock in strata (13-30, 36-40) versus pollock in the deep strata (23-24, 27-30, 36-38).

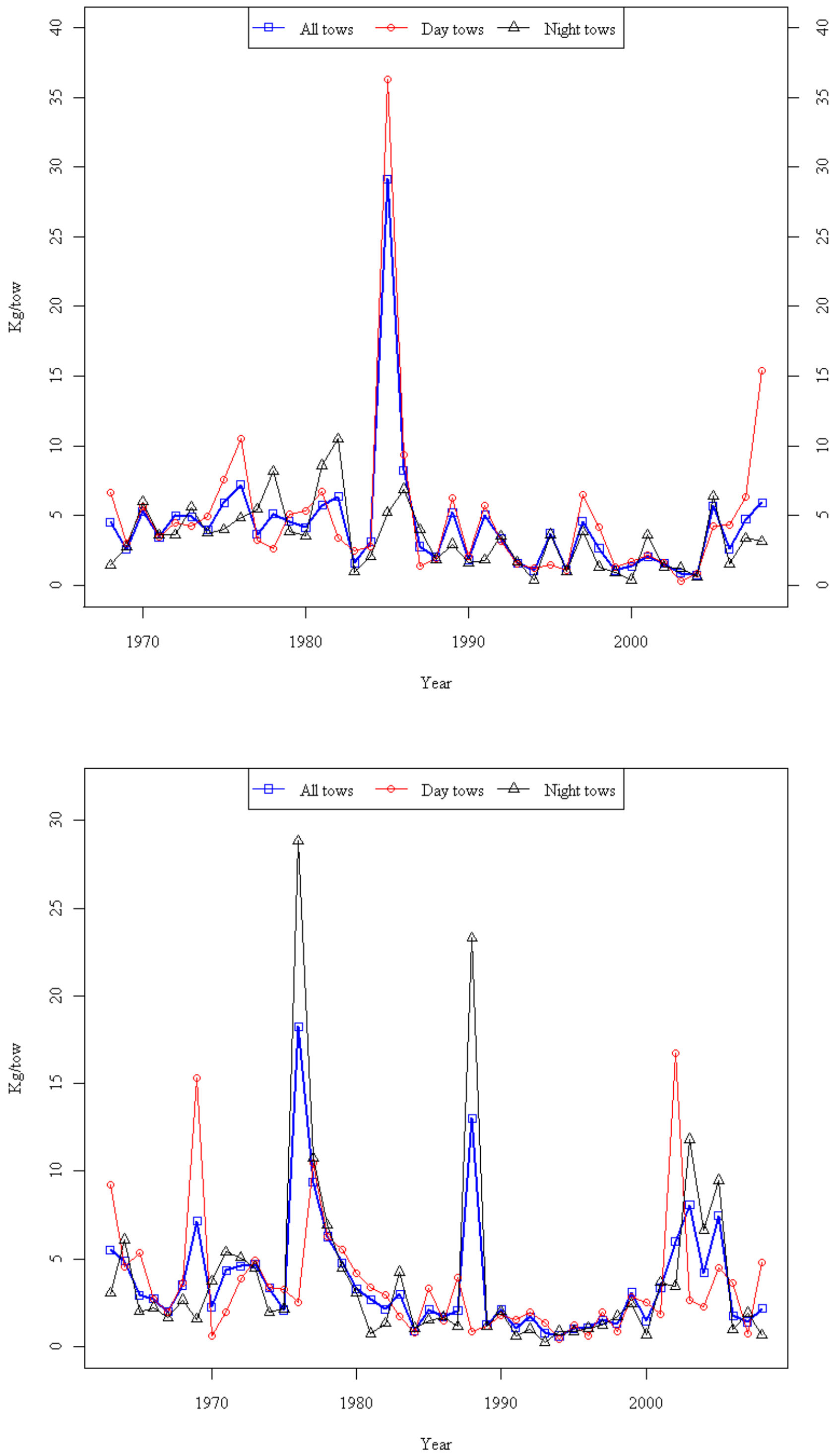


Figure C20. Comparison of NEFSC bottom trawl survey indices for Pollock in the spring (top) or fall (bottom). In blue is the index using all tows, while daylight tows are plotted in red and night tows are plotted in black.

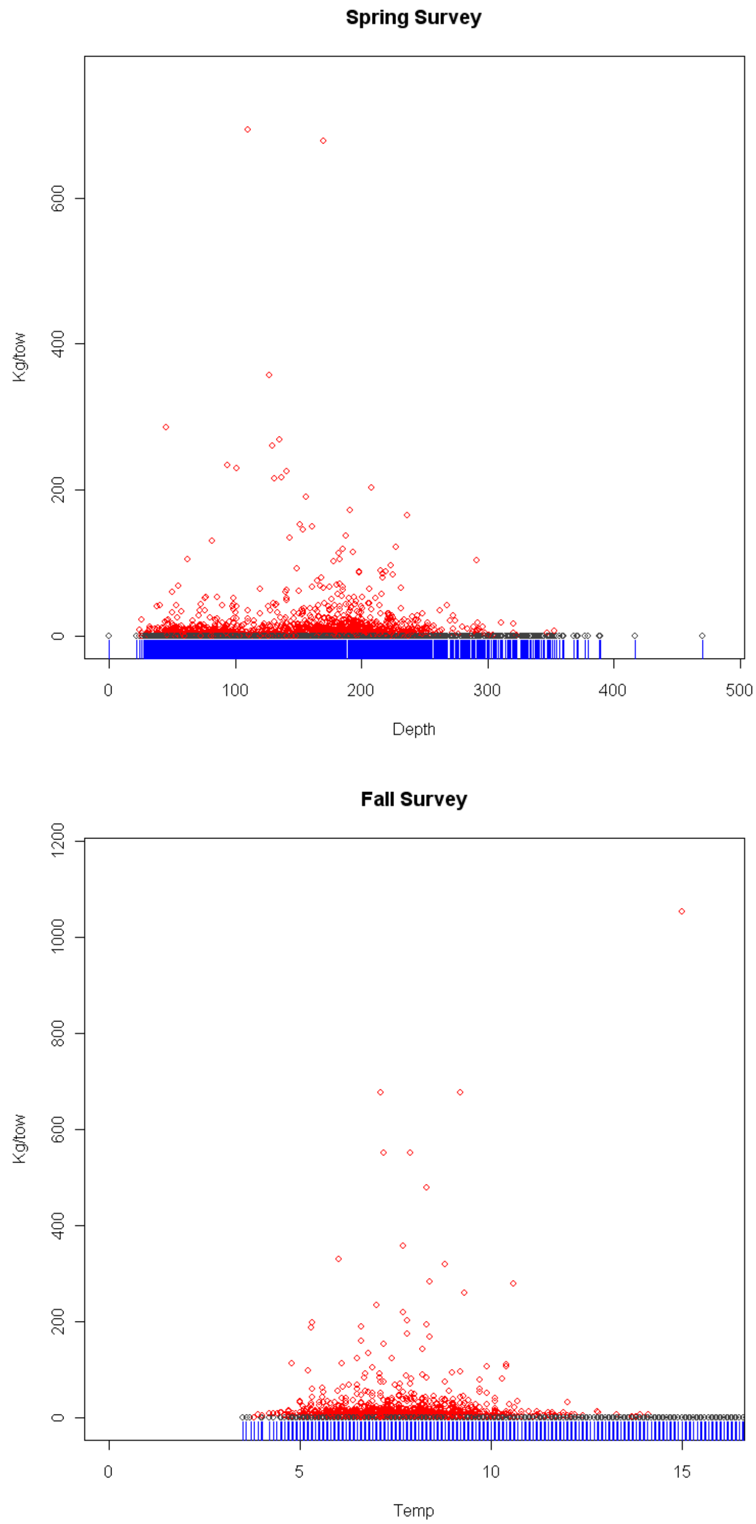


Figure C21. Plot of bottom temperature on a given tow and the corresponding kg/tow of Pollock. Red circles are nonzero, black circles are zero tows, and the blue vertical lines are a 'rug plot' to indicate the number of observations at a given temperature.

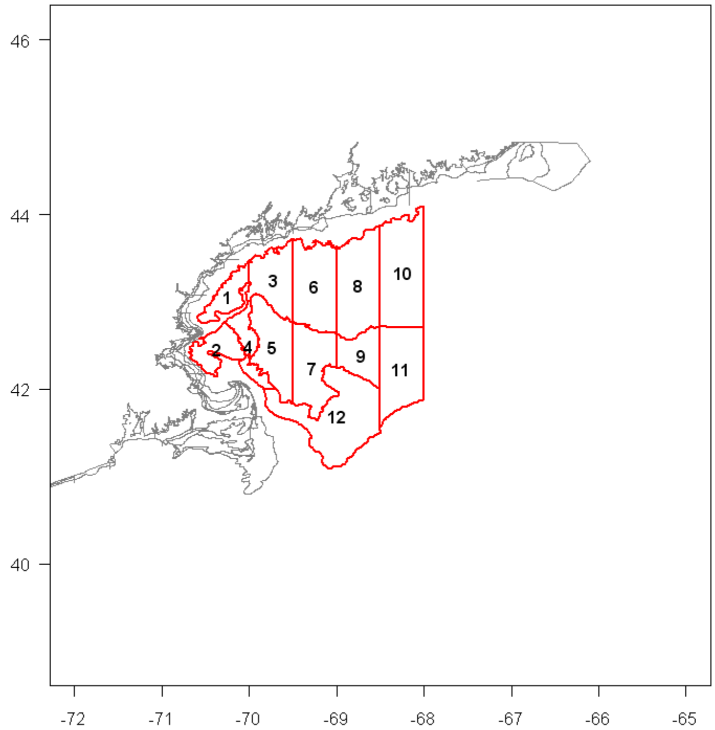


Figure C22. NEFSC summer survey strata in the Gulf of Maine.

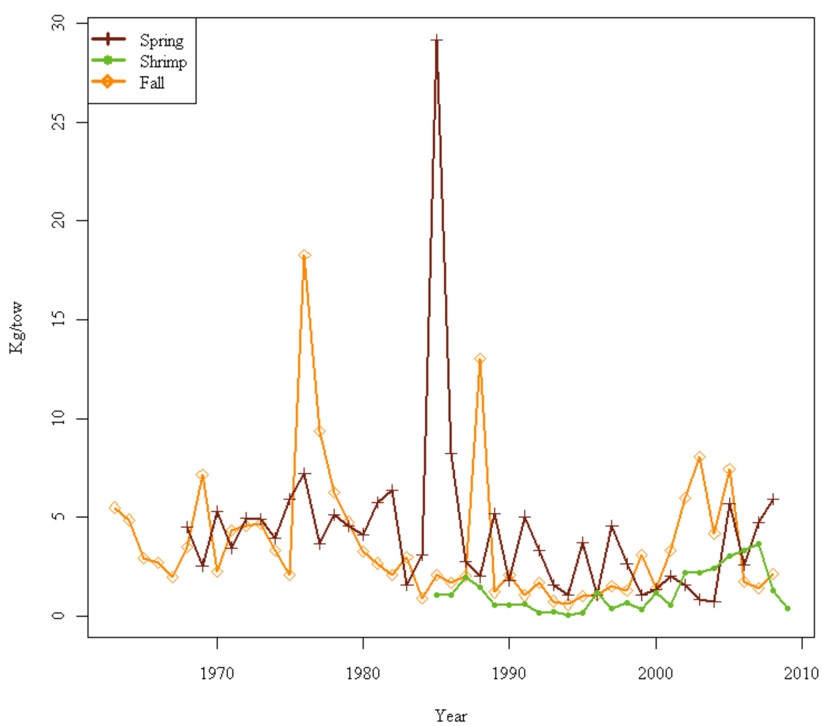
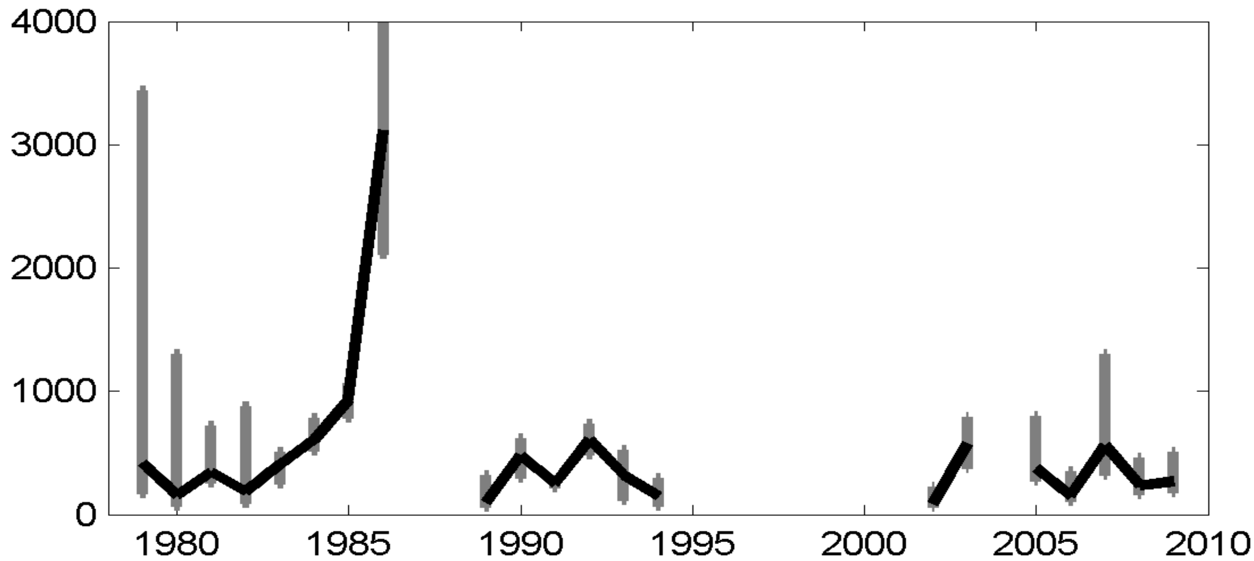
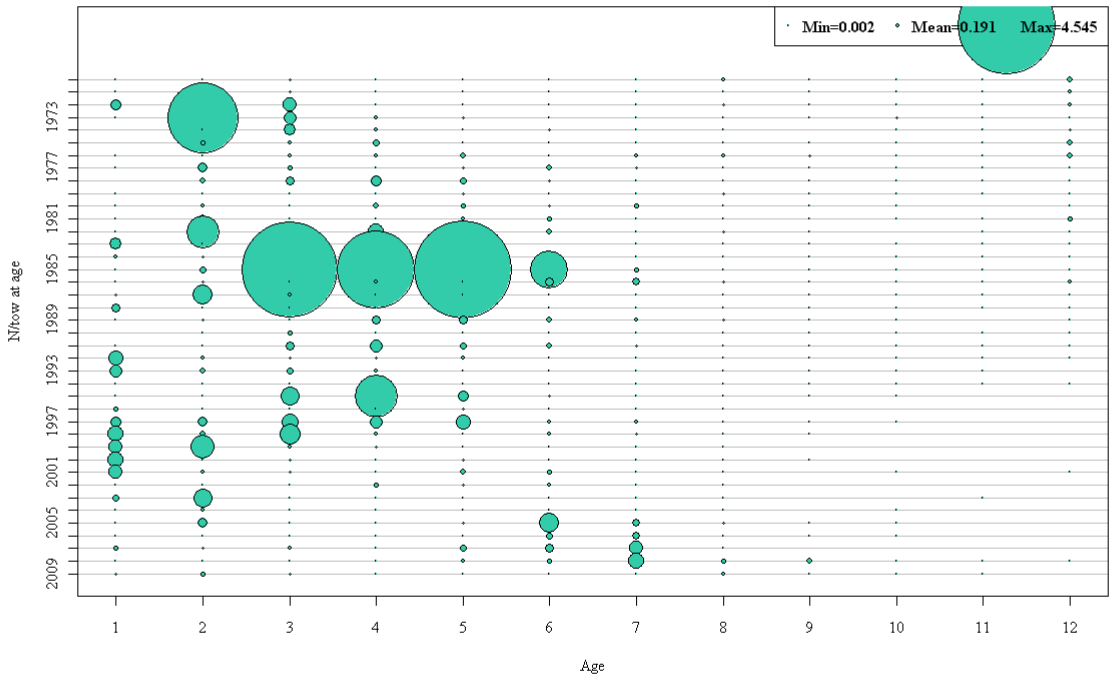


Figure C23. NEFSC fall, spring and summer survey indices for pollock.



C24. Larval index for pollock from ichthyoplankton data, which could be used as an index of spawning biomass. Units are number per 10m^2.



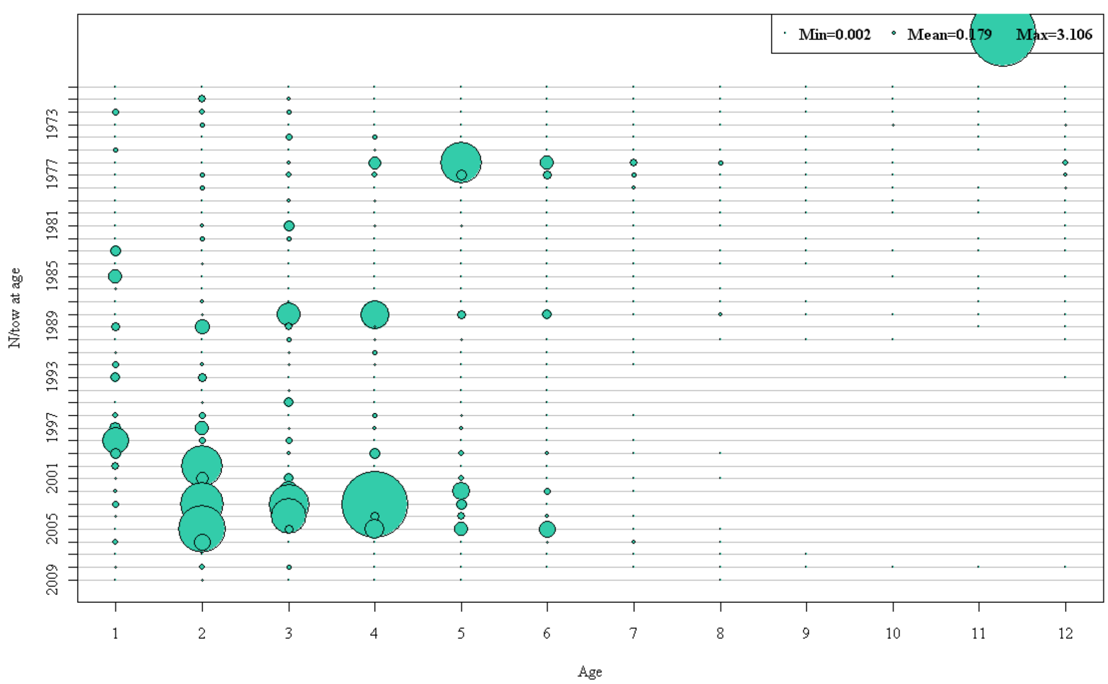


Figure C25. Survey age structure in the NEFSC spring (top) and the NEFSC fall (bottom).

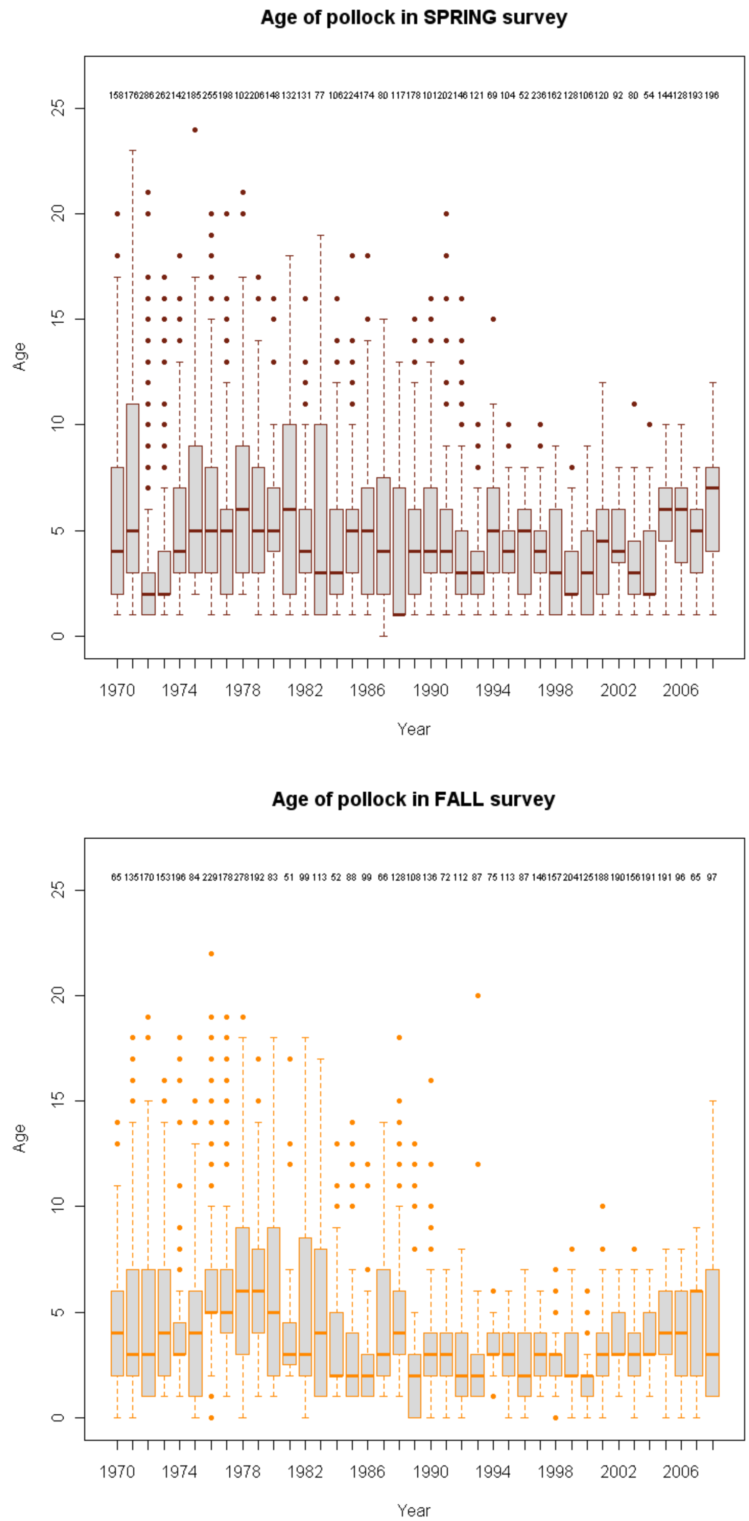
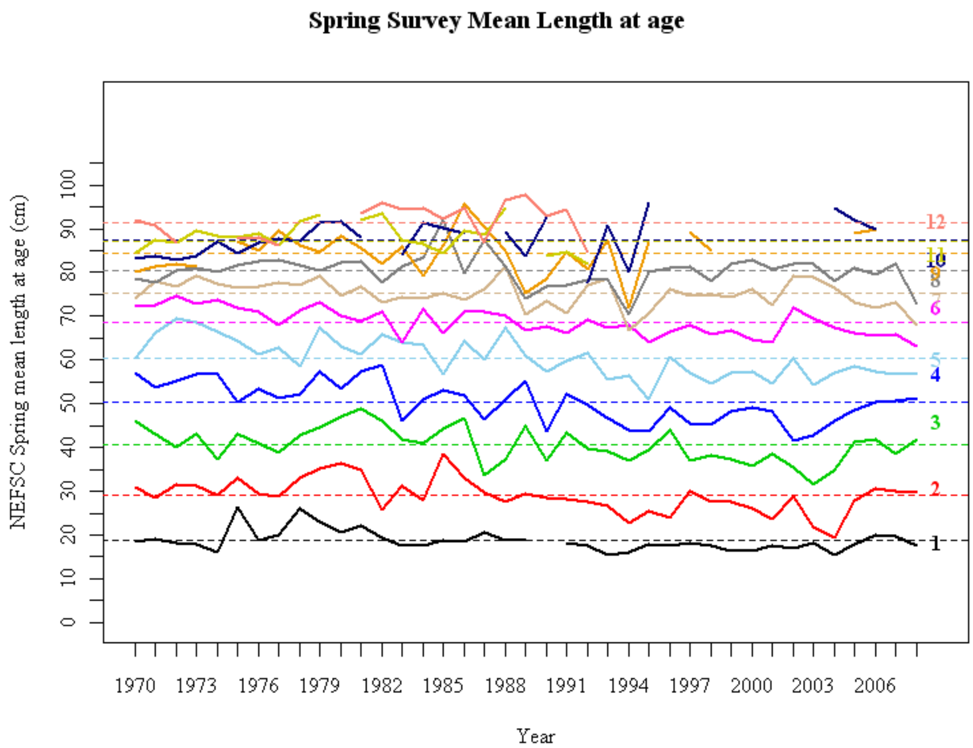


Figure C26. Annual box-plot of NEFSC bottom trawl spring and fall survey age structure.



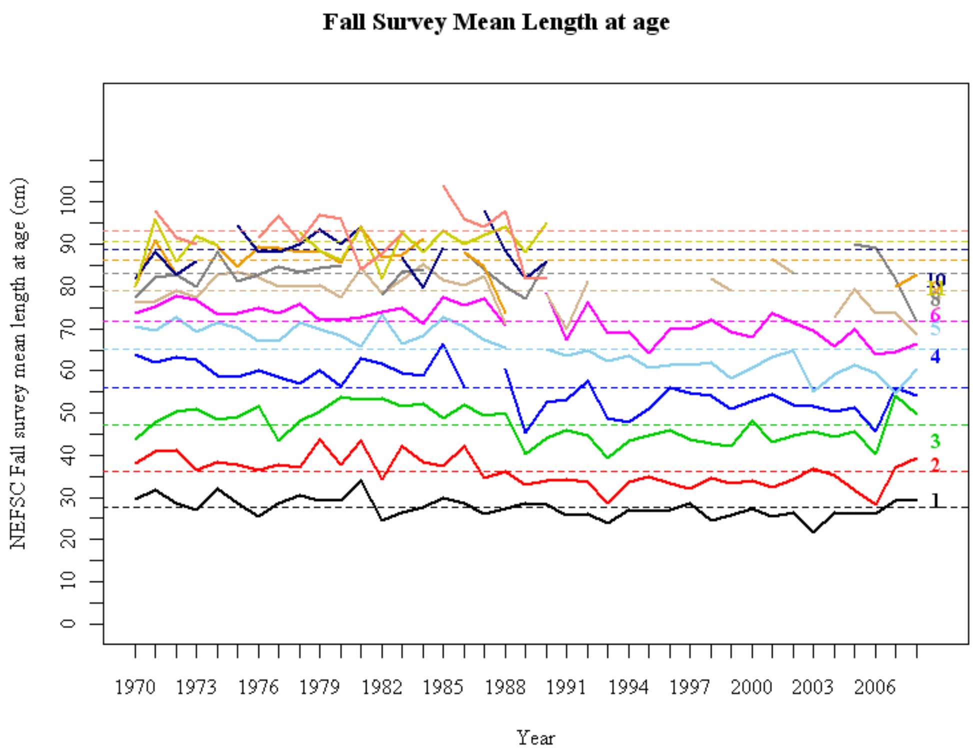


Figure C27. Mean size at age (cm) of pollock from length samples in the NEFSC bottom trawl spring and fall surveys. For each age, the time series mean length is plotted with a dashed line in the same color as the mean length trend.

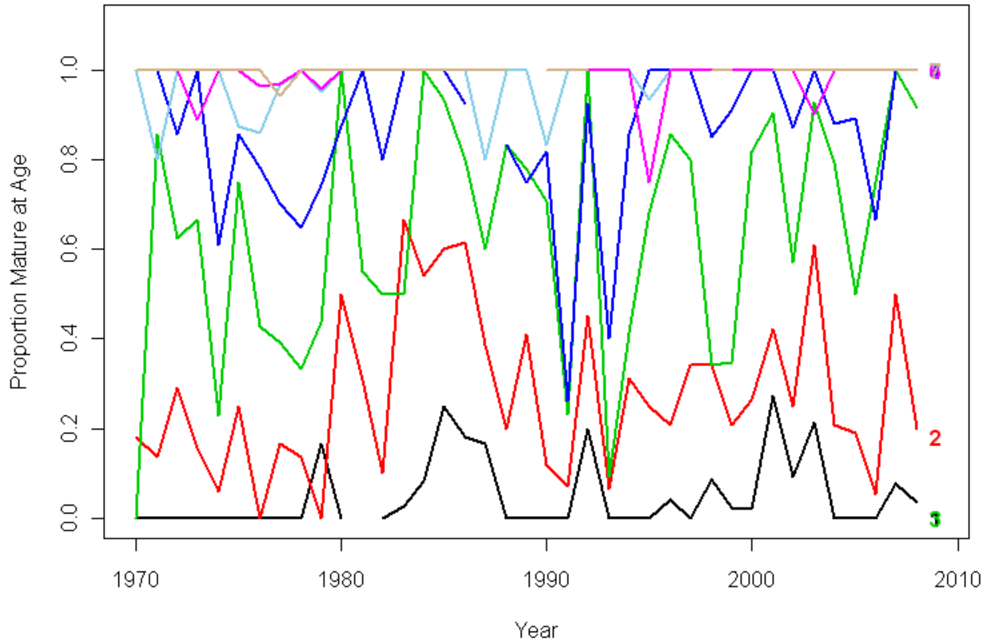


Figure C28. Pollock maturity at age by year from samples in the NEFSC fall bottom trawl survey.

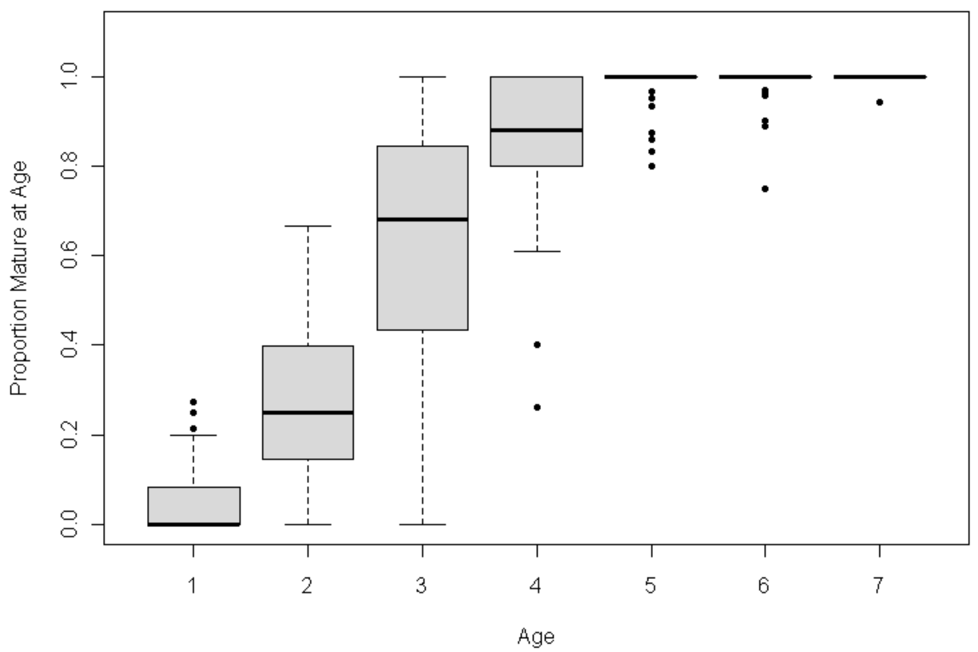
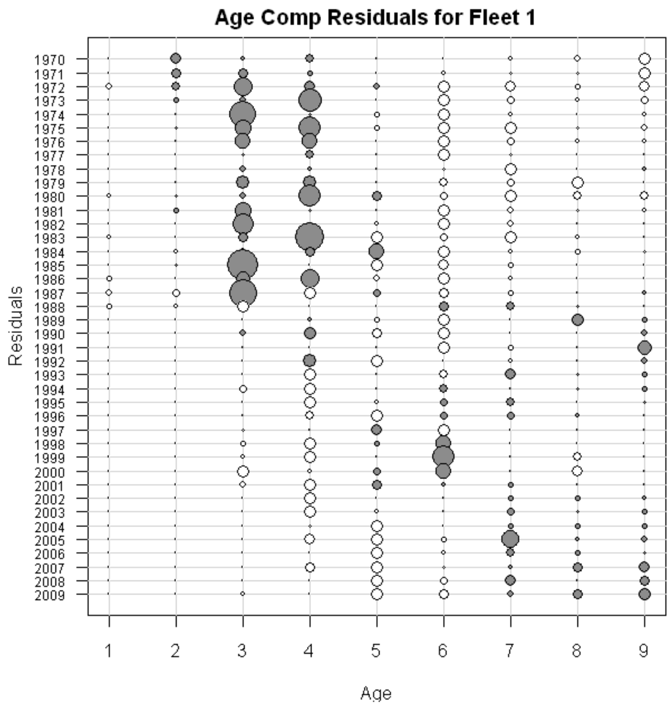


Figure C29. Pollock maturity at age, pooled across all years, from samples in the NEFSC fall bottom trawl survey.



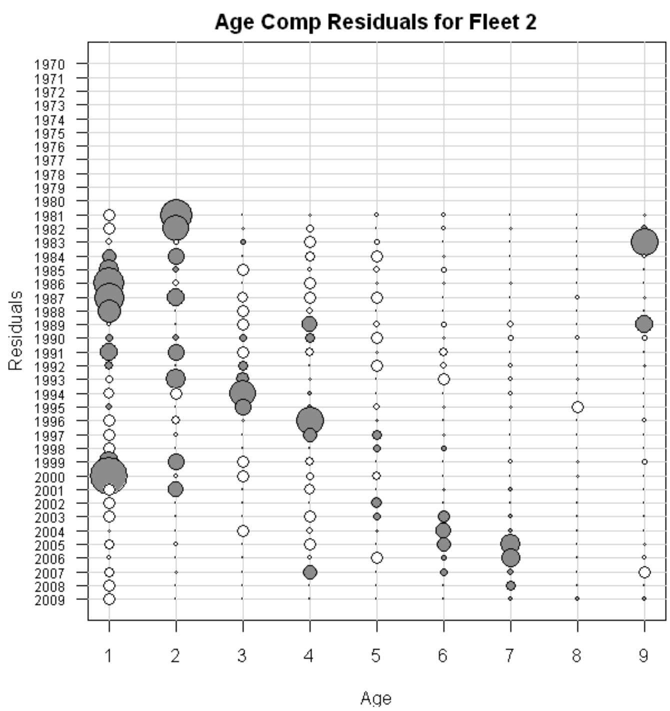


Figure C30. Residuals (predicted-observed) for age composition in the commercial (Fleet 1) and recreational (Fleet 2) fishery when only 1 selectivity block is used for each fleet in the ASAP base model. This was an exploratory model, and the residual pattern supports the addition of more selectivity blocks.

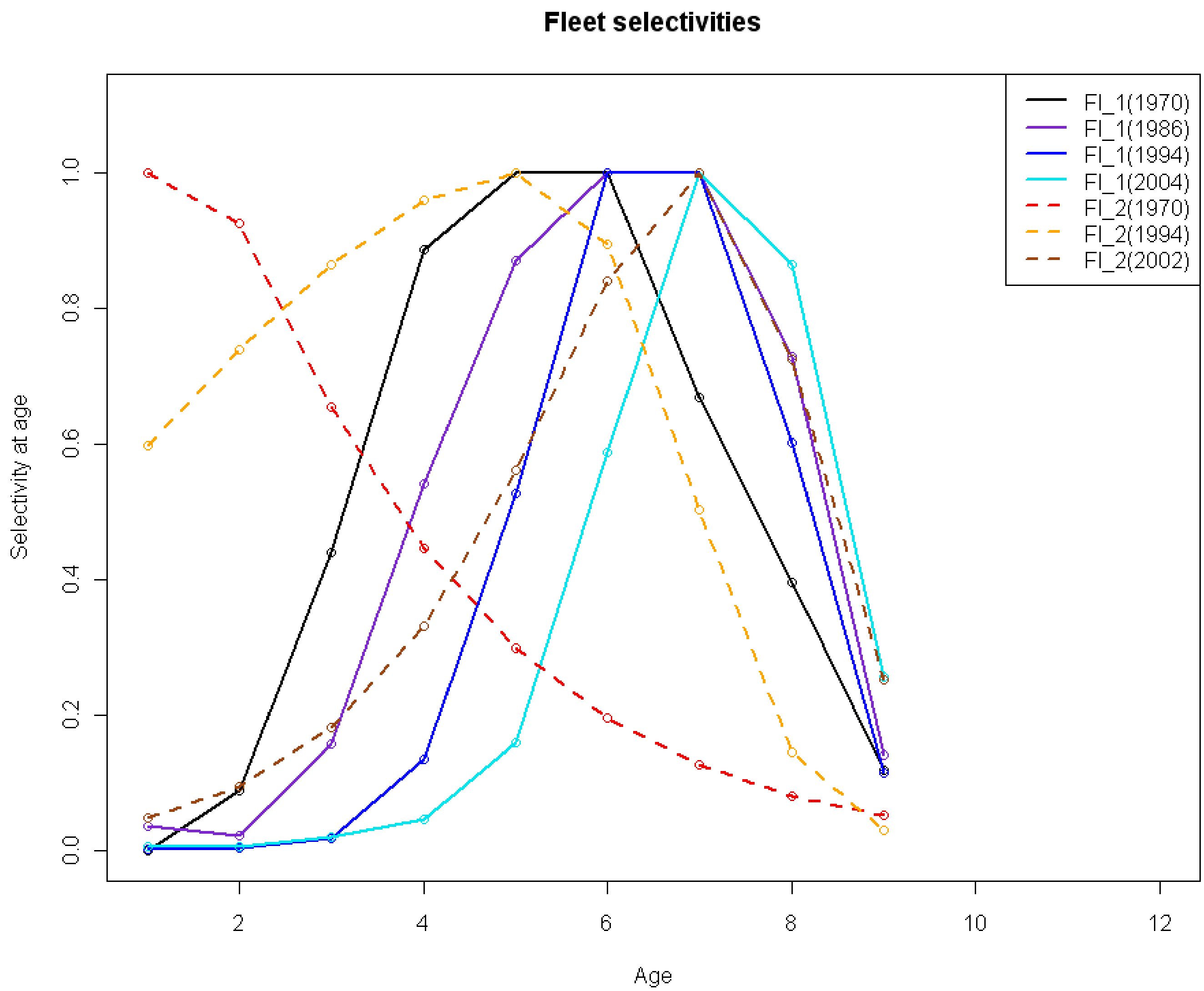


Figure C31. Selectivity blocks estimated for each fleet in the ASAP base model (solid lines for commercial, dashed lines for recreational). The legend indentifies either the commercial (Fl_1) or recreational (Fl_2) fleet, and in parentheses are the first years that each new selectivity vector was used.

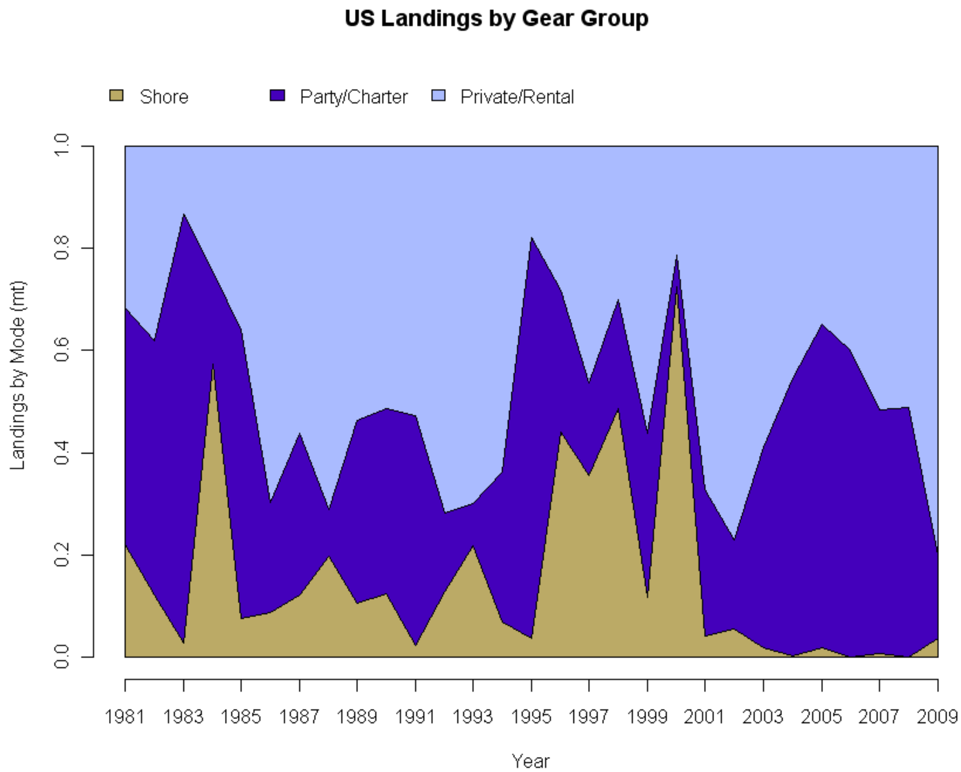


Figure C32. Proportional composition of recreational landings by mode.

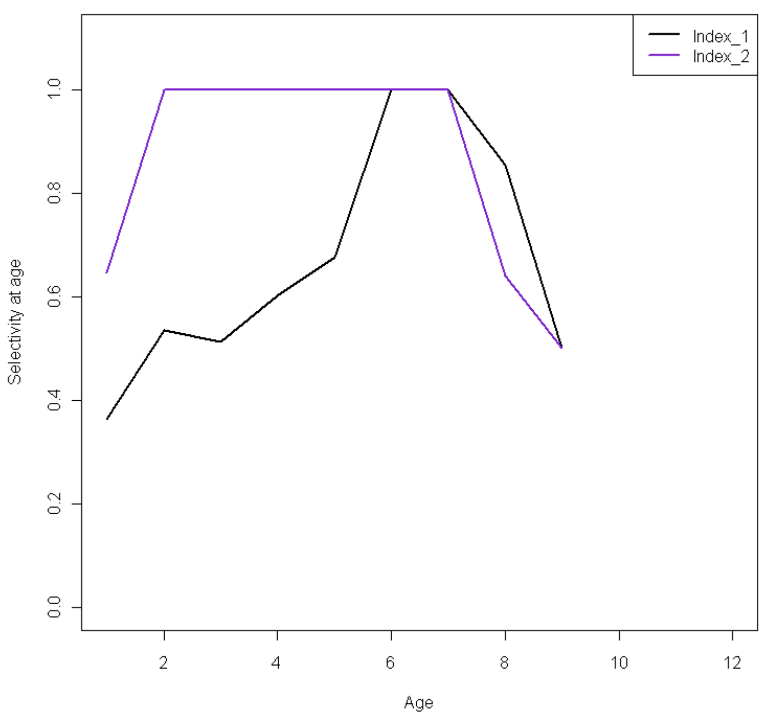


Figure C33. Selectivity at age for the NEFSC spring (Index_1) and fall (Index_2) surveys from the ASAP base model.

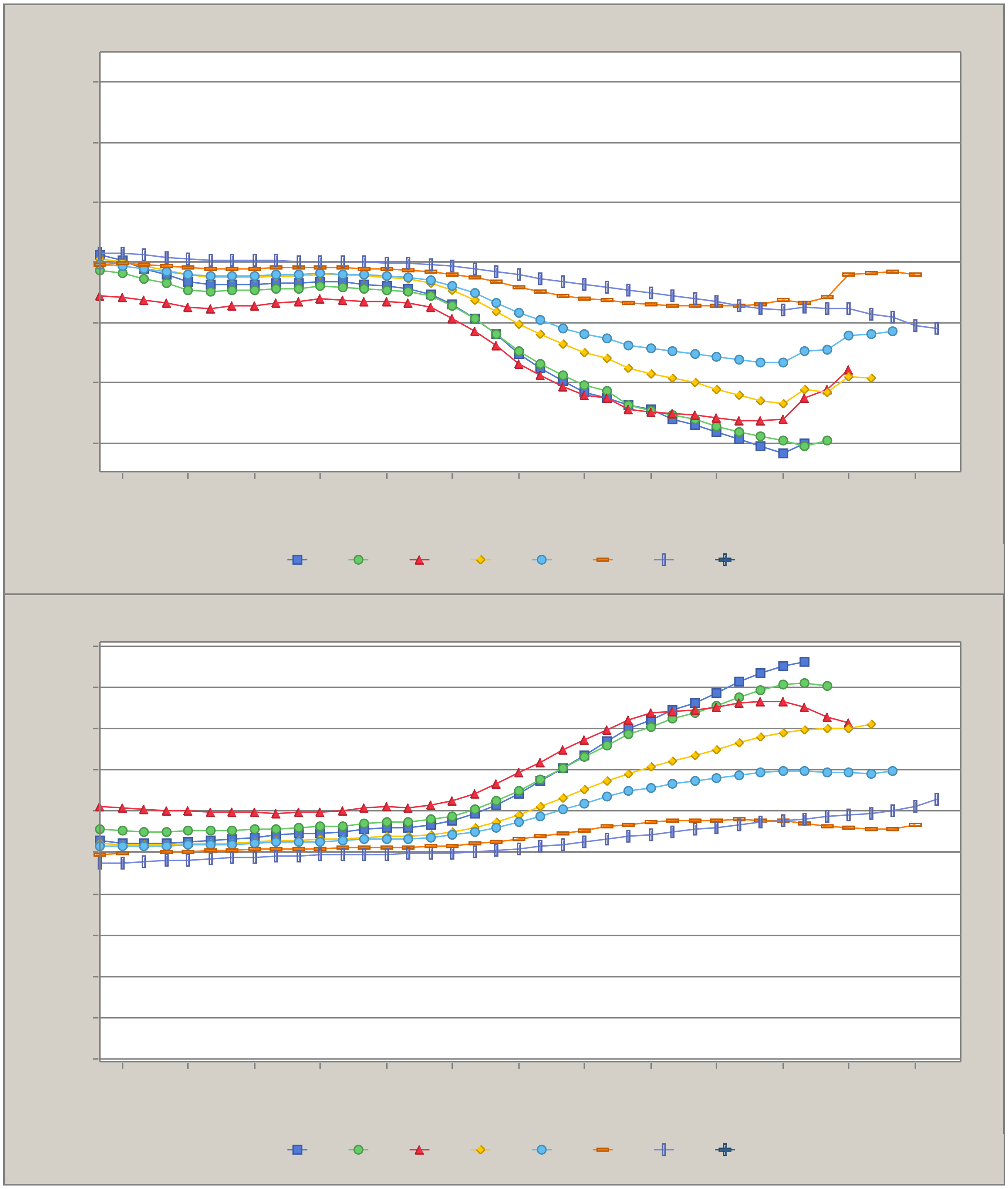


Figure C34. Retrospective analysis for the ASAP base model for years 2002-2008.

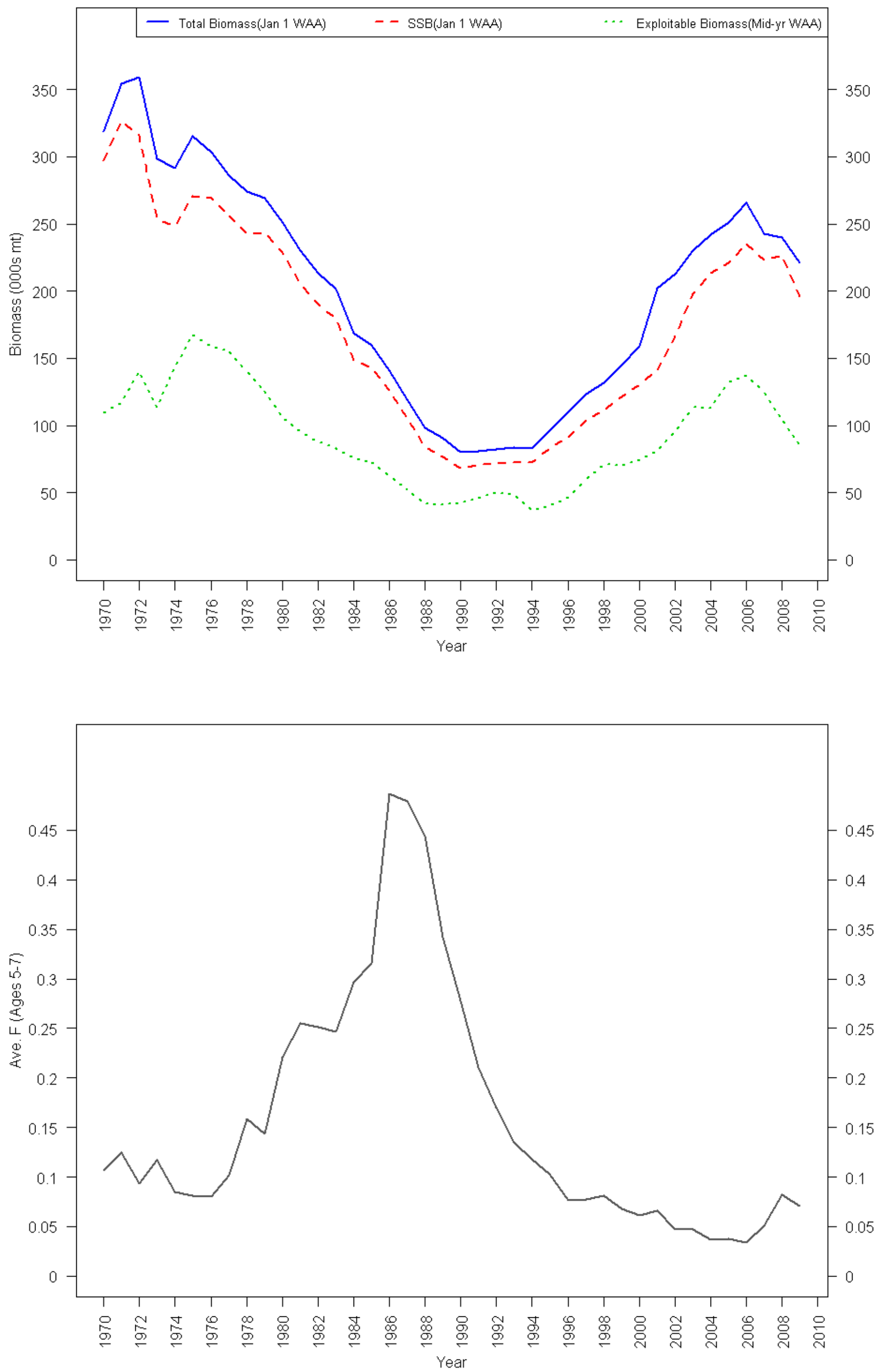


Figure C35. Annual estimates biomass (mt) and $F_{5\text{-}7}$ from the ASAP base model.

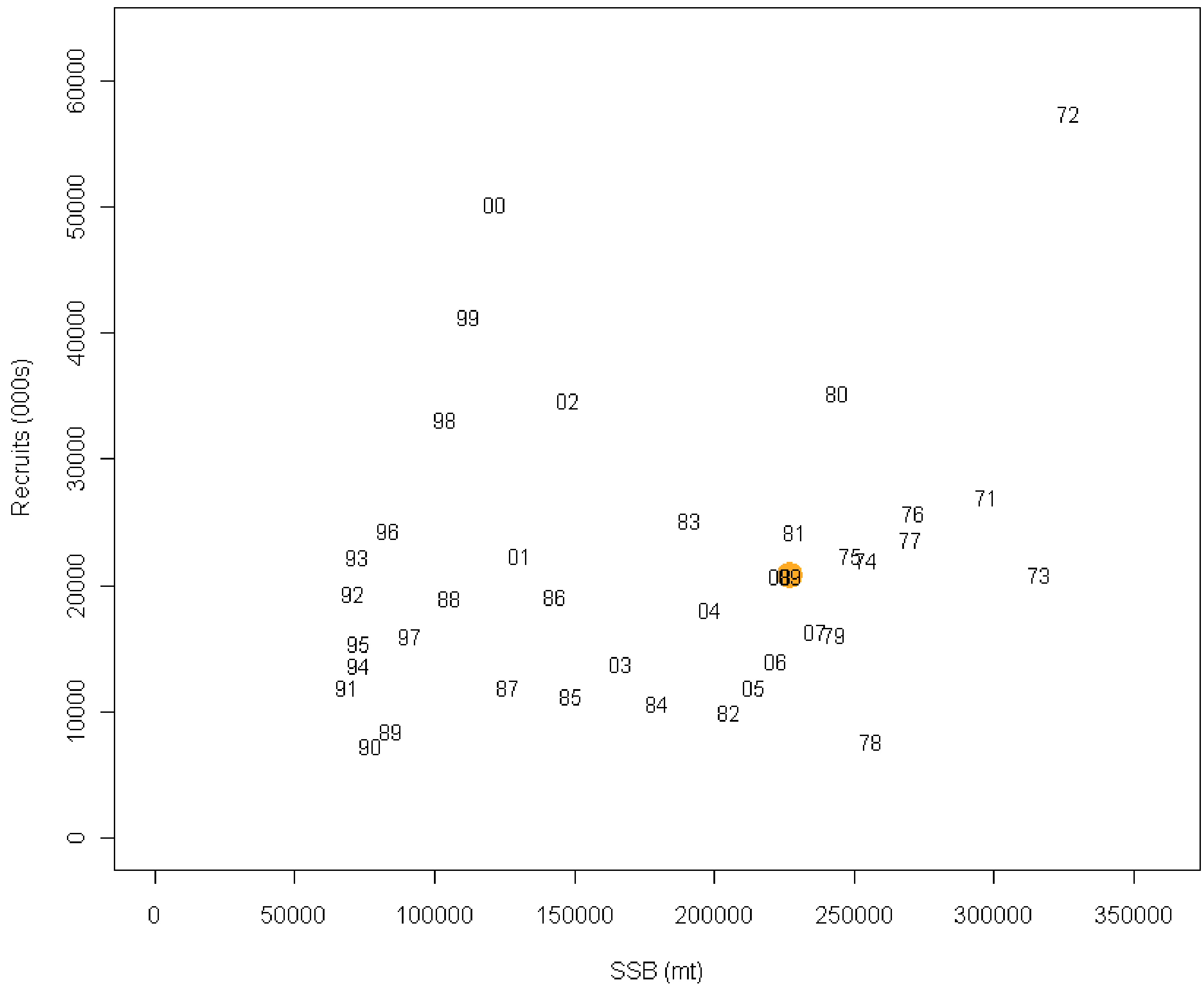


Figure C36. Scatterplot of ASAP estimates of spawning stock biomass (SSB, mt) versus recruitment at age 1 (thousands of fish). The symbol for each observation is the last two digits of the year (e.g. '09' is the model estimate of age 1 recruitment in year 2009). The most recent recruitment estimate for 2009 is highlighted by a filled orange circle.

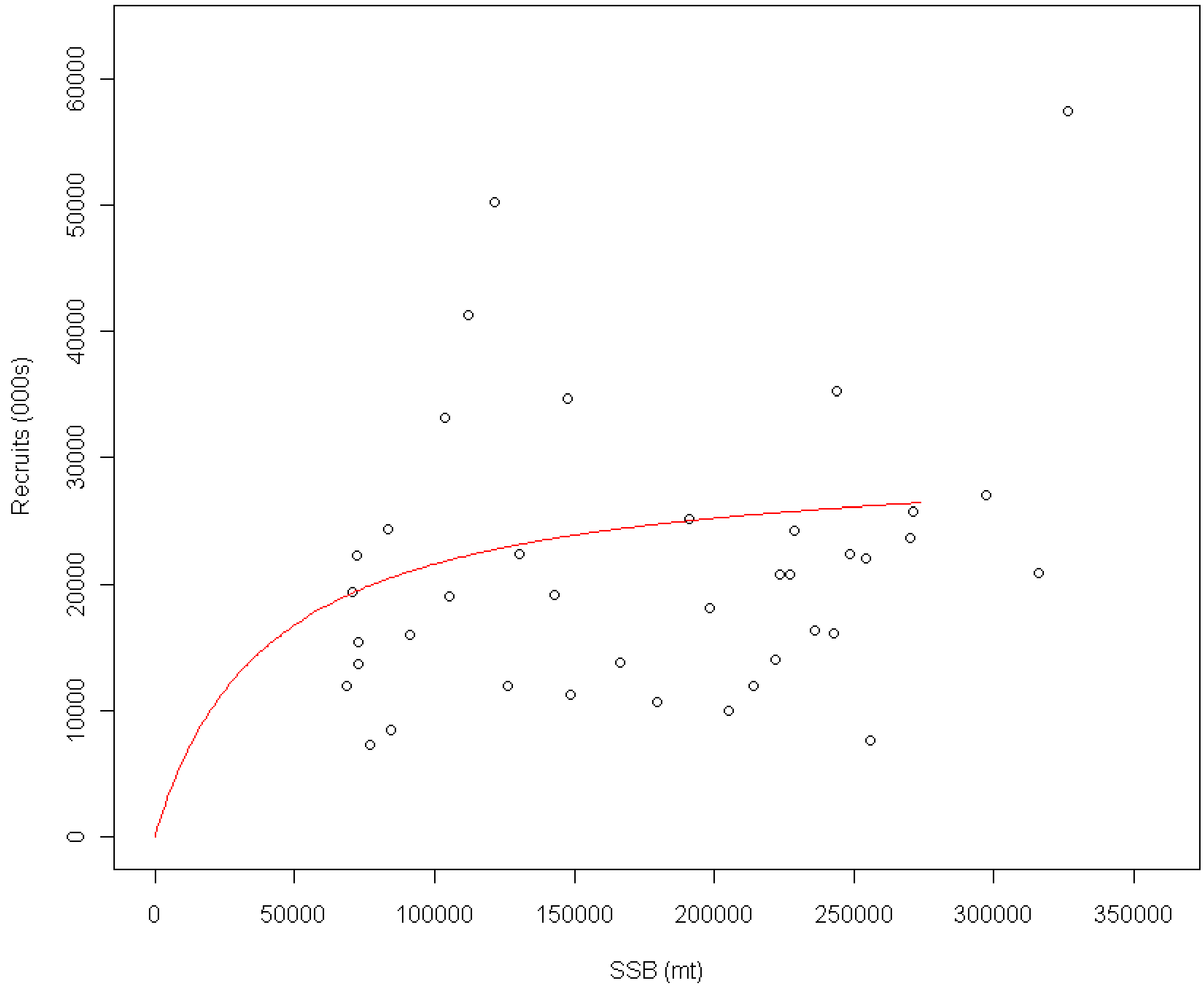


Figure C37. ASAP base model of the predicted stock recruit relationship (solid red line) and the estimated spawning stock biomass (SSB mt) and age 1 recruits (in thousands of fish).

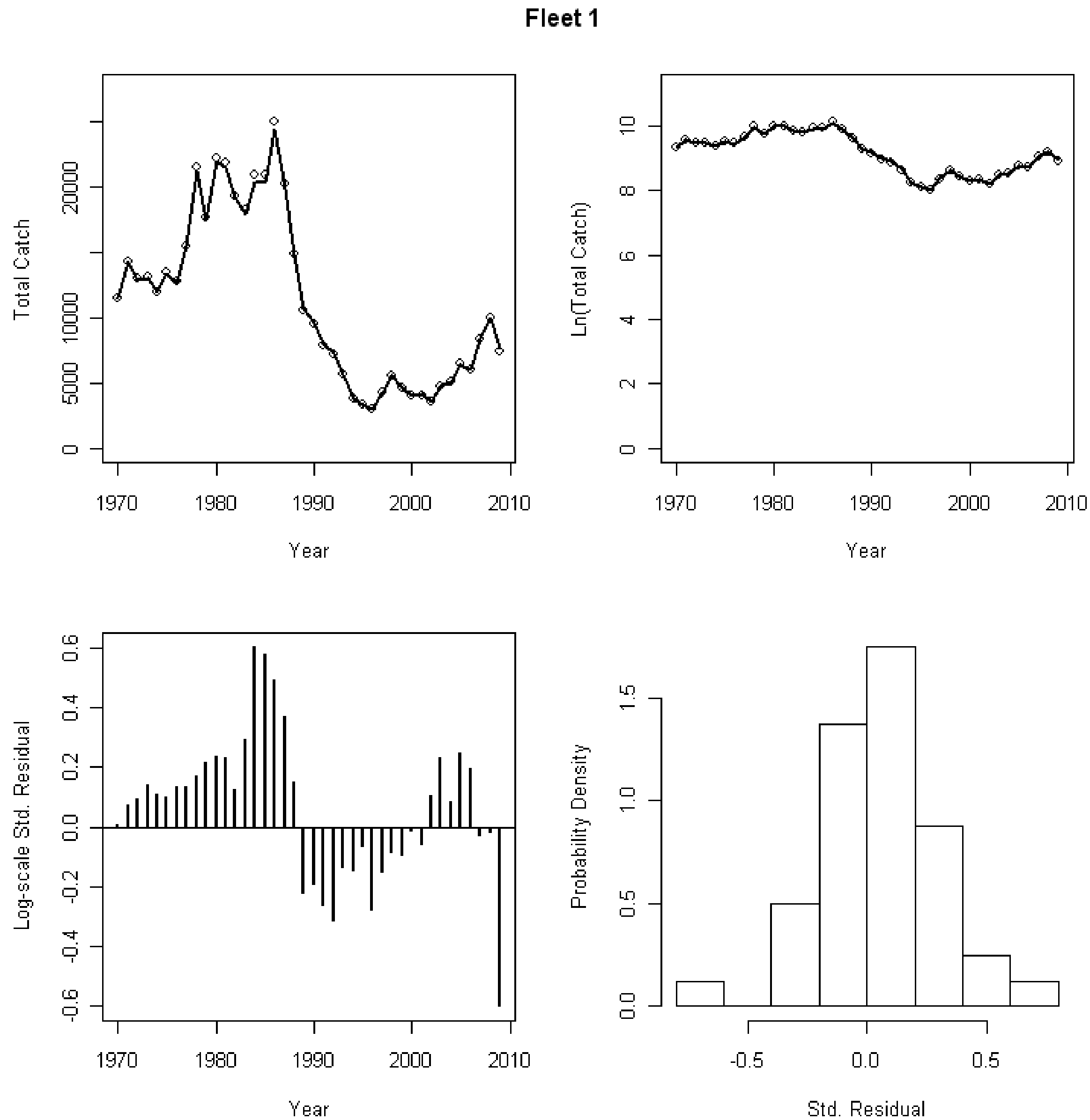


Figure C38. ASAP base model fit to commercial landings.

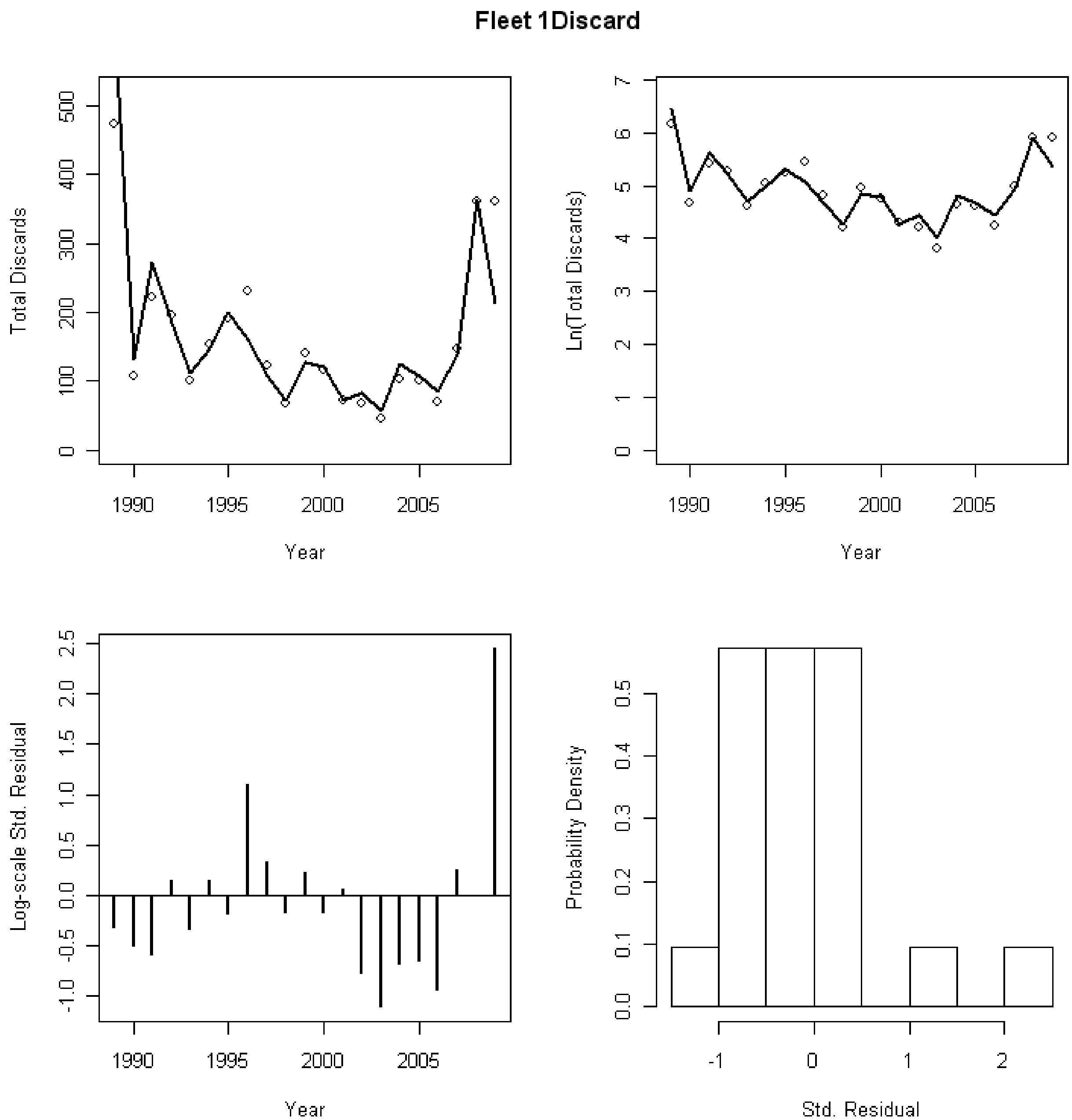


Figure C39. ASAP base model fit to commercial discards.

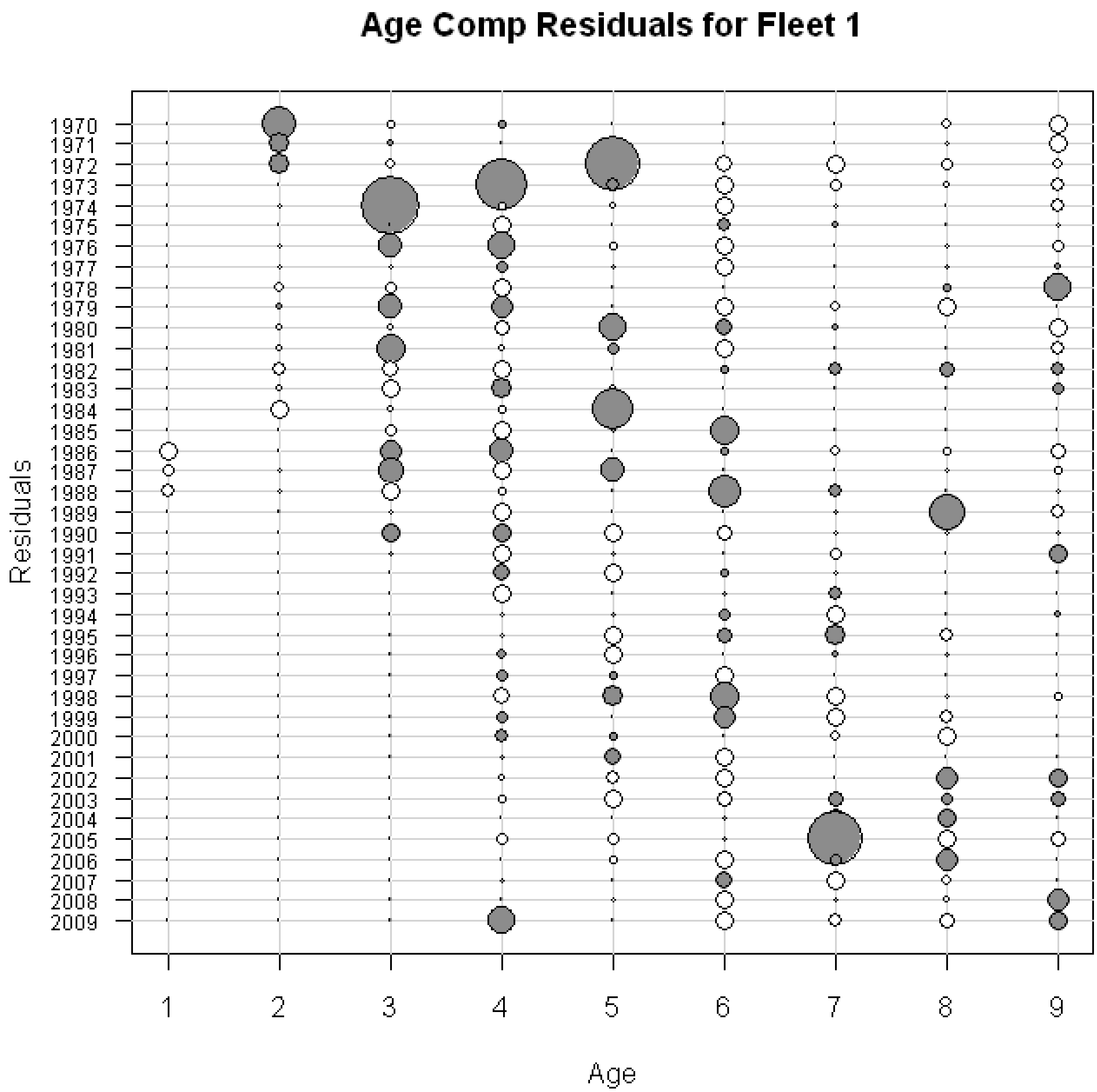


Figure C40. ASAP base model residuals for commercial catch age composition. Open circles are positive residuals, filled circles are negative residuals, calculated as (Predicted-Observed).

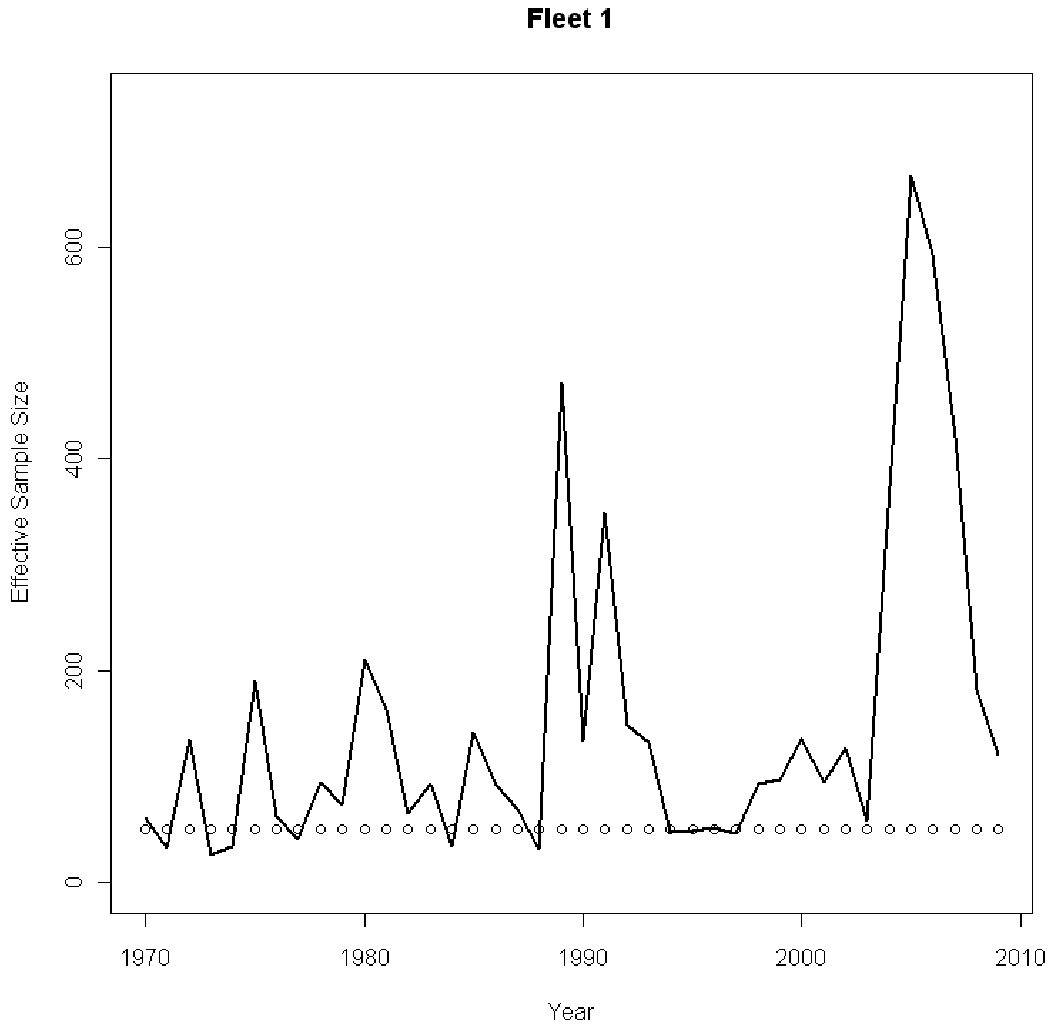


Figure C41. ASAP base model comparison of input effective sample size versus the model estimated effective sample size for the commercial fleet.

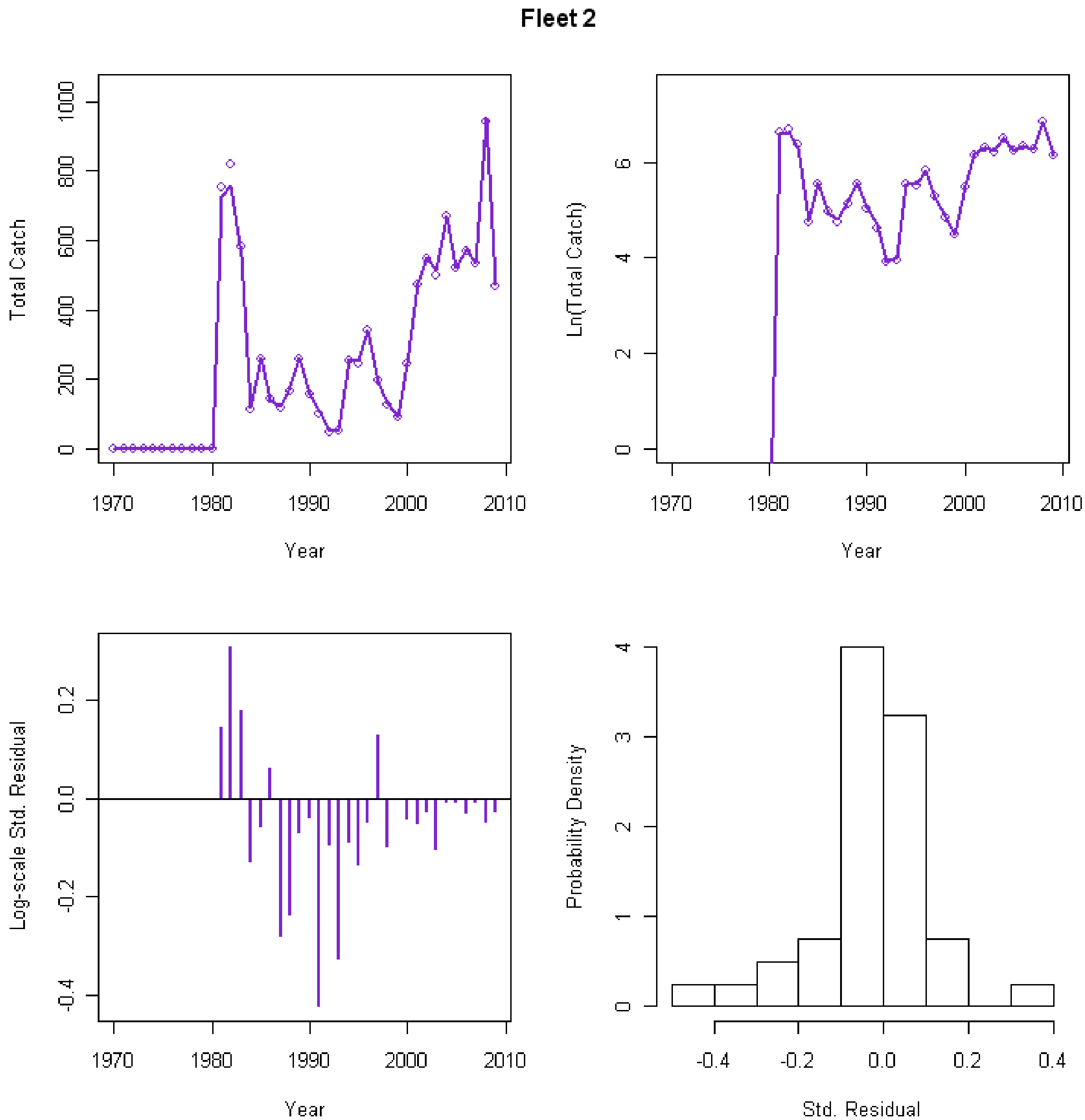


Figure C42. ASAP base model fit to recreational landings.

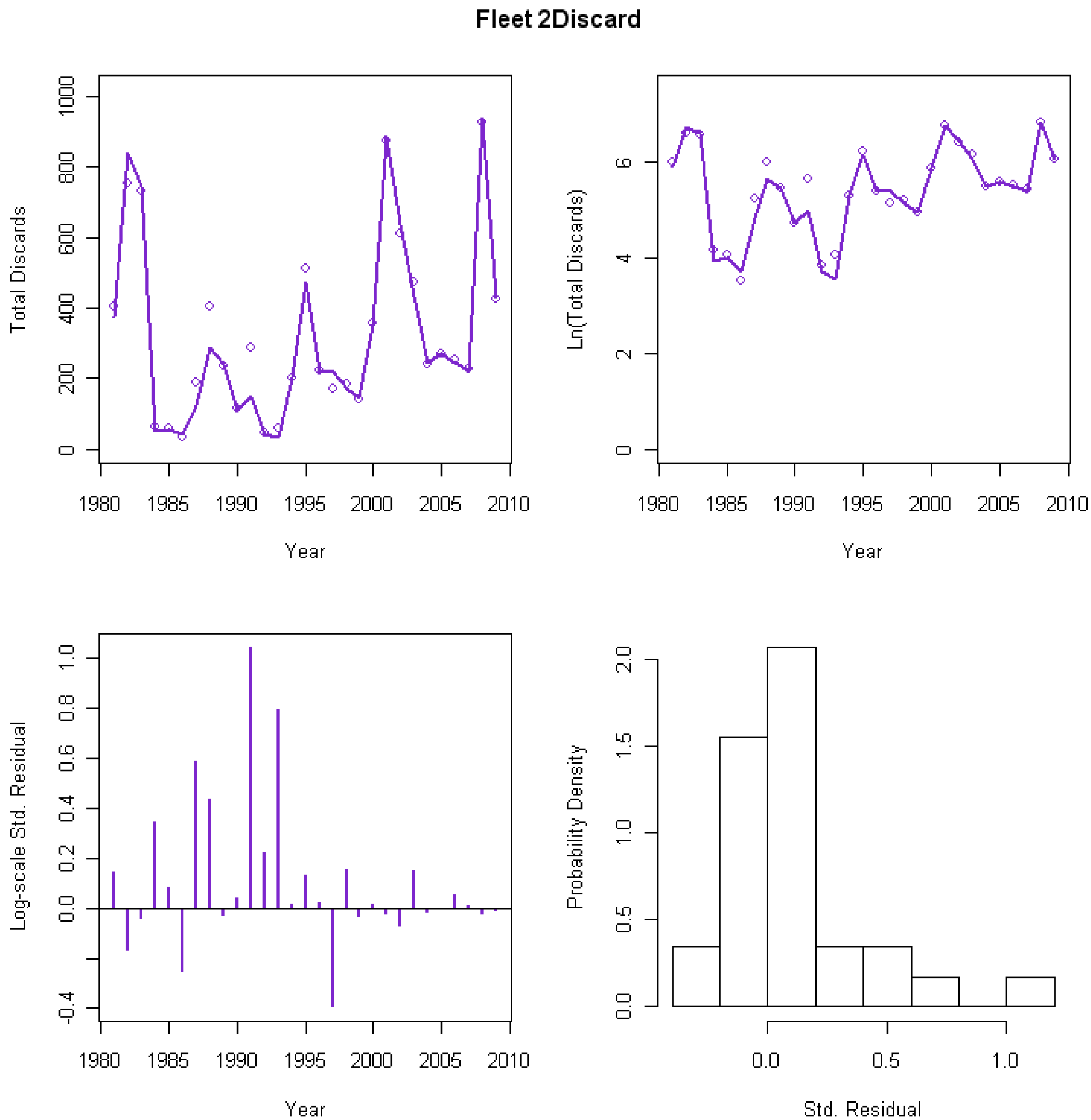


Figure C43. ASAP base model fit to recreational discards.

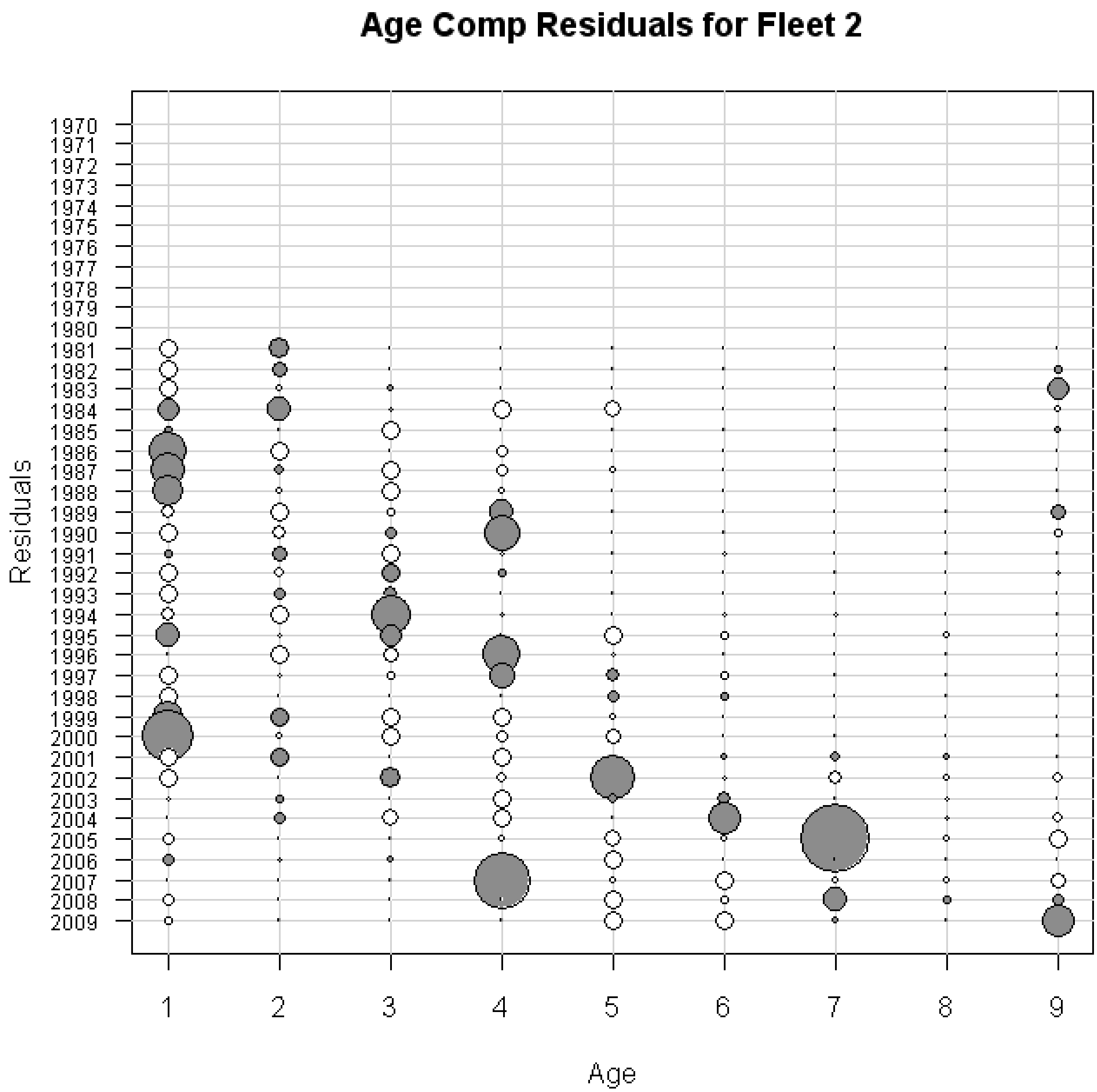


Figure C44. ASAP base model residuals for recreational catch age composition. Open circles are positive residuals, filled circles are negative residuals, calculated as (Predicted-Observed).

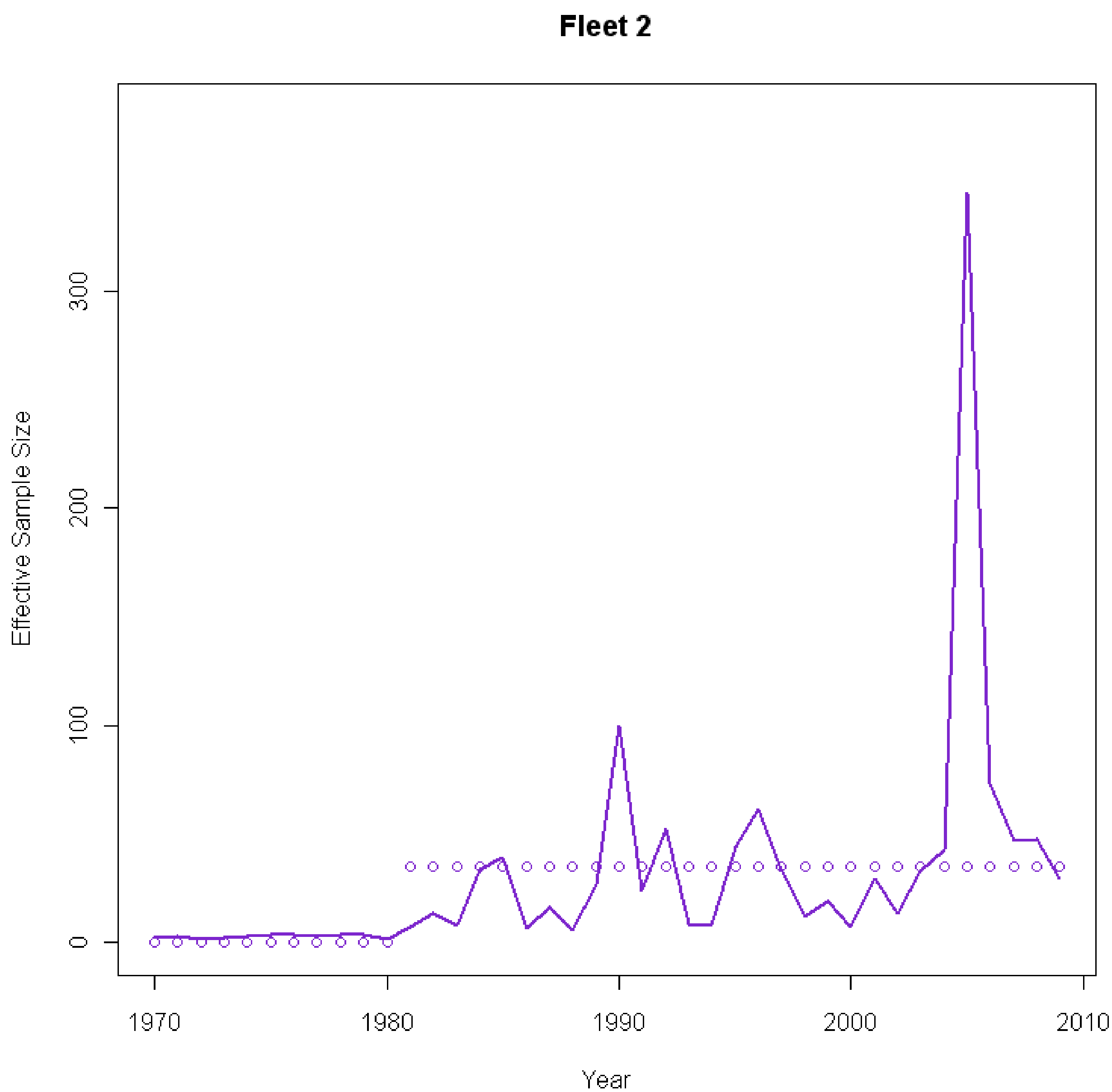


Figure C45. ASAP base model comparison of input effective sample size versus the model estimated effective sample size for the recreational fleet.

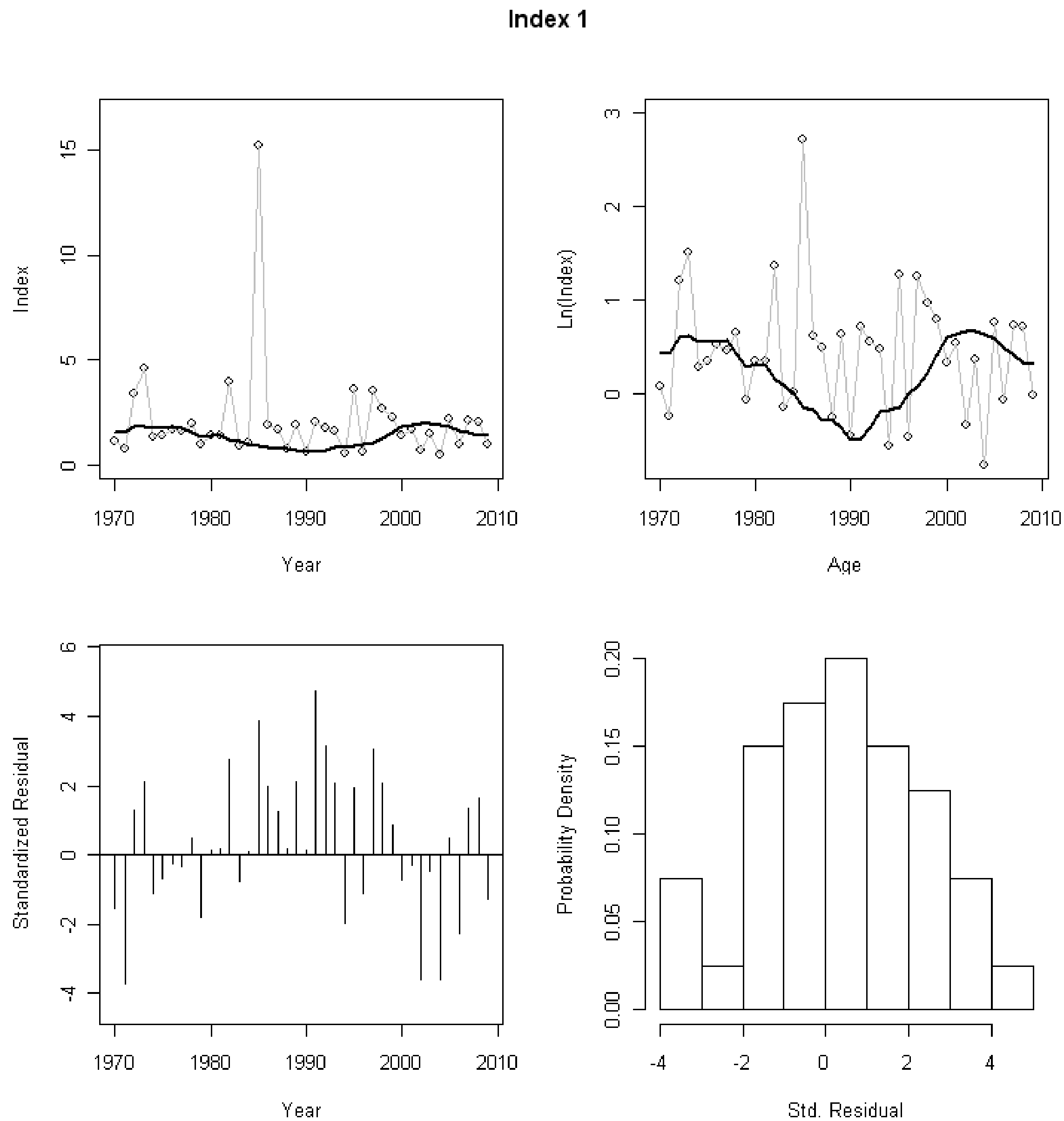


Figure C46. ASAP base model fit to the NEFSC spring index.

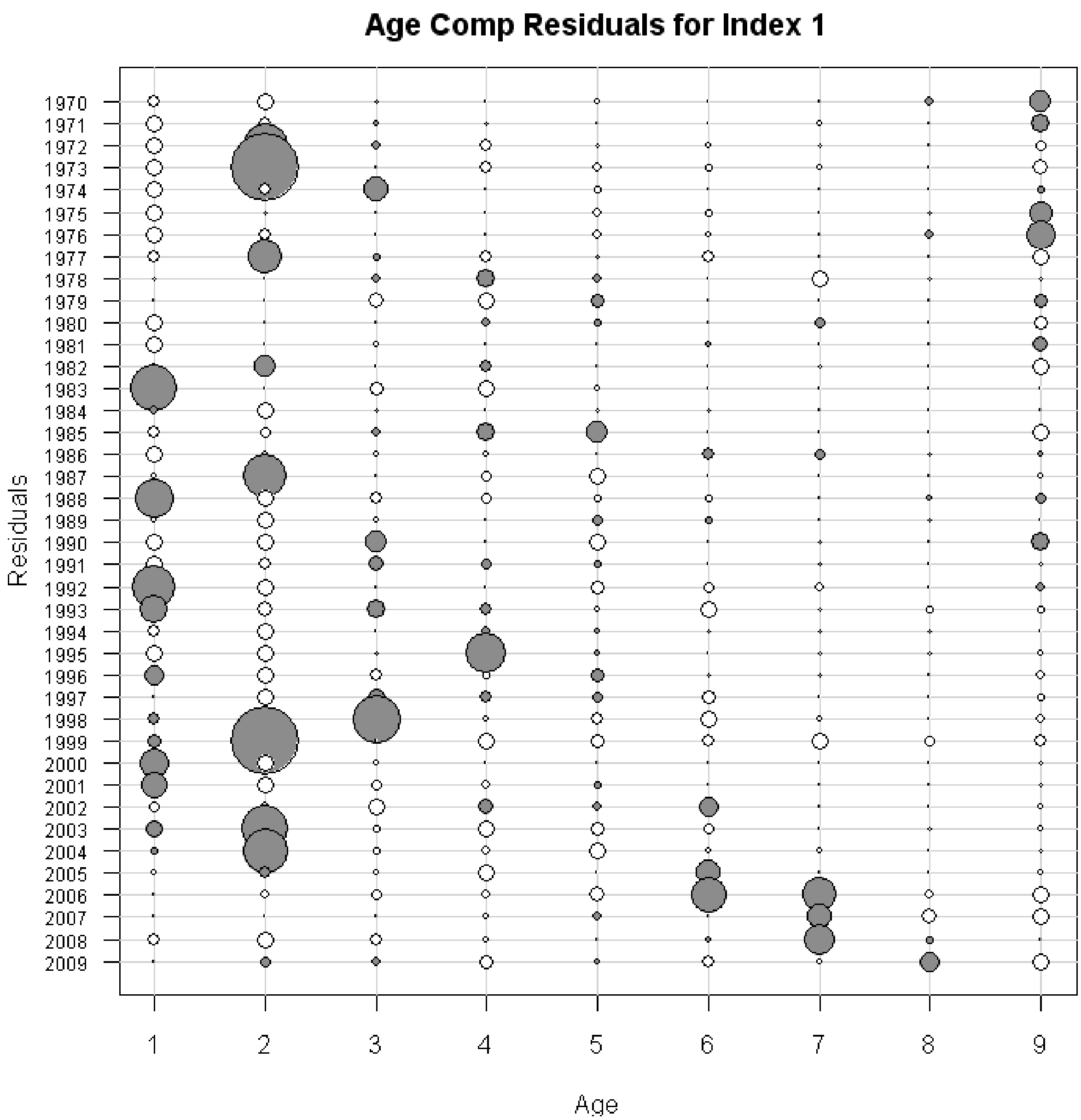


Figure C47. ASAP base model residuals for NEFSC spring index age composition. Open circles are positive residuals, filled circles are negative residuals, calculated as (Predicted-Observed).

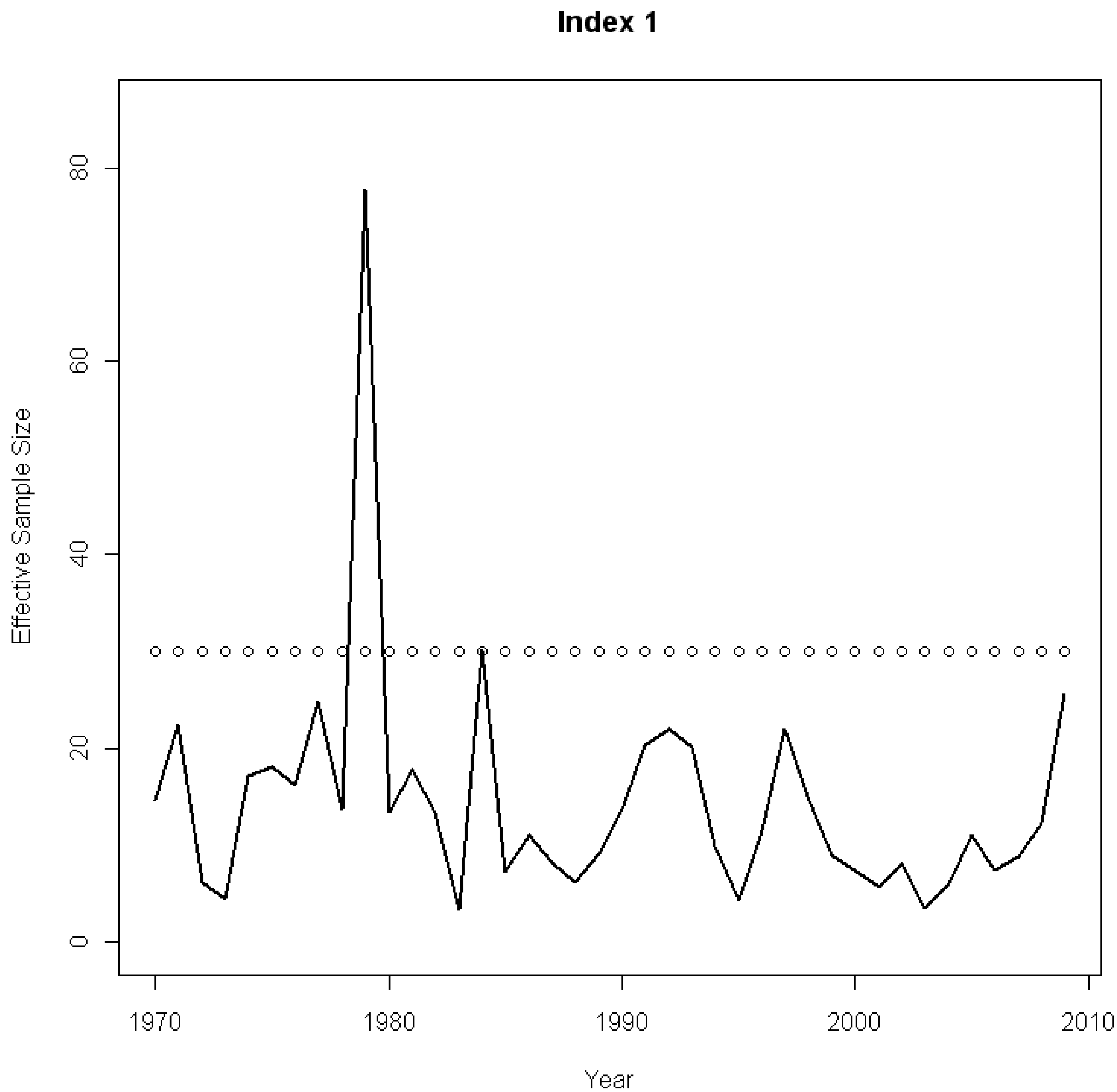


Figure C48. ASAP base model comparison of input effective sample size versus the model estimated effective sample size for the NEFSC spring index.

Index 2

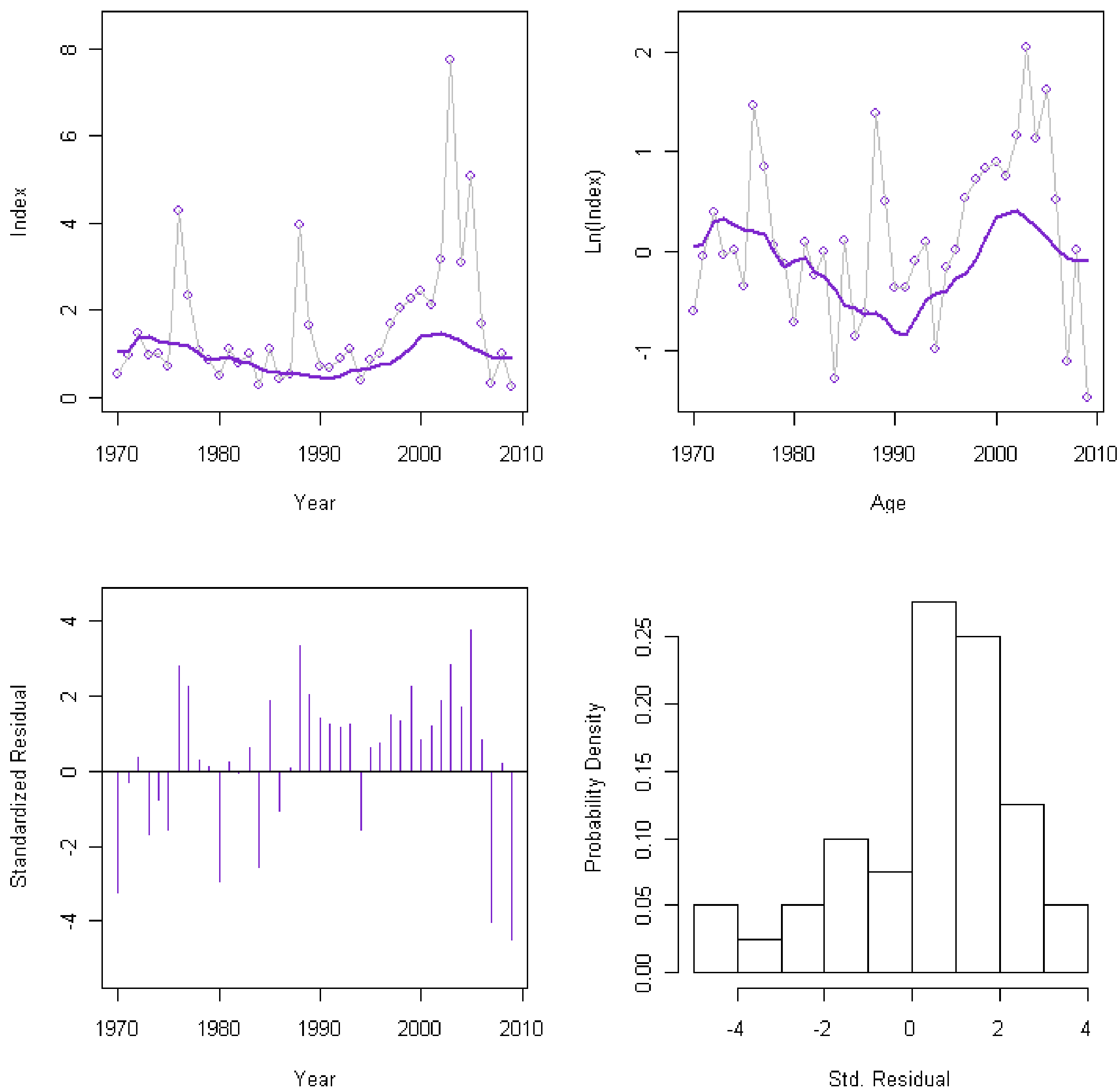


Figure C49. ASAP base model fit to the NEFSC fall index.

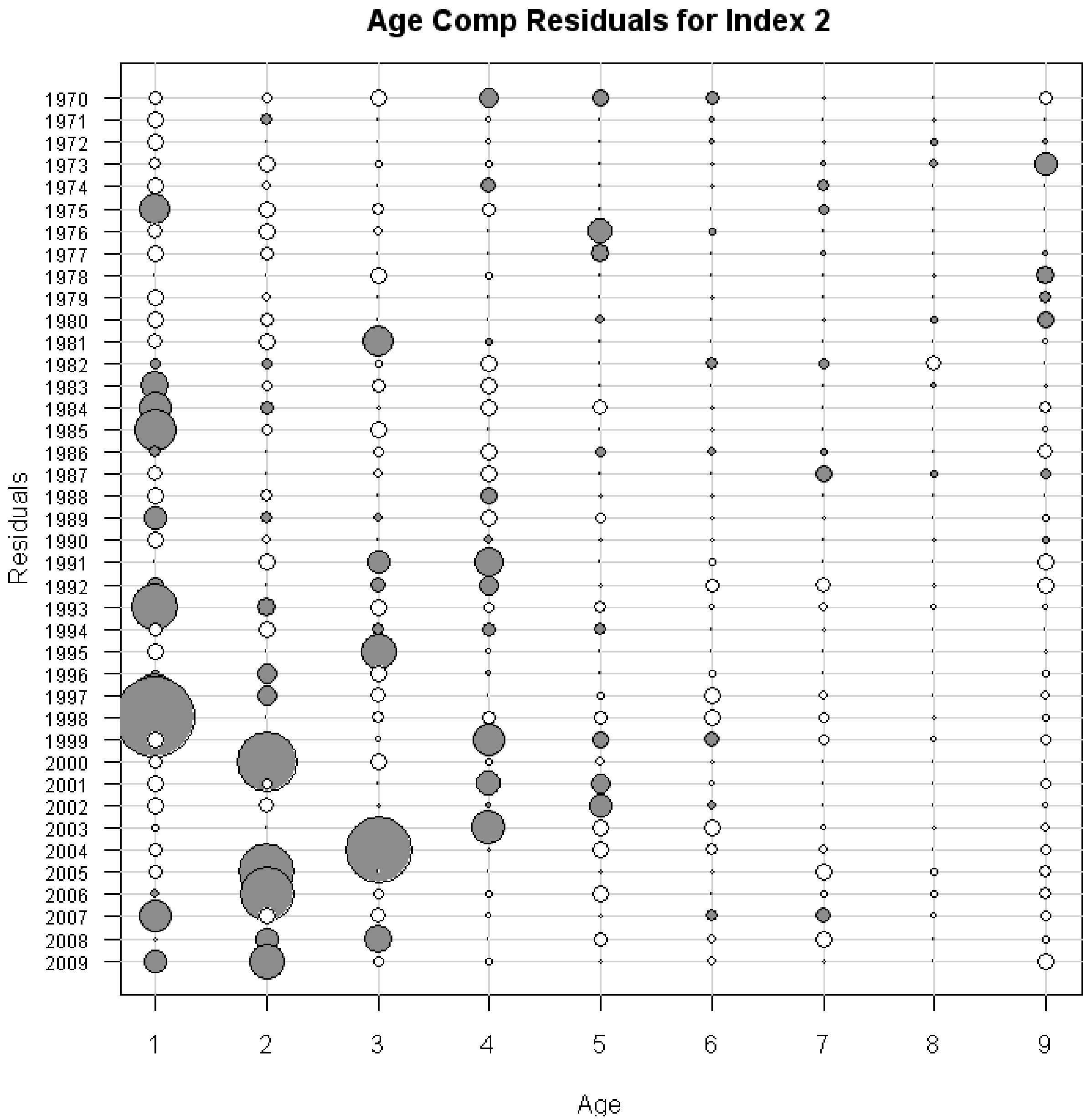


Figure C50. ASAP base model residuals for NEFSC fall index age composition. Open circles are positive residuals, filled circles are negative residuals, calculated as (Predicted-Observed).

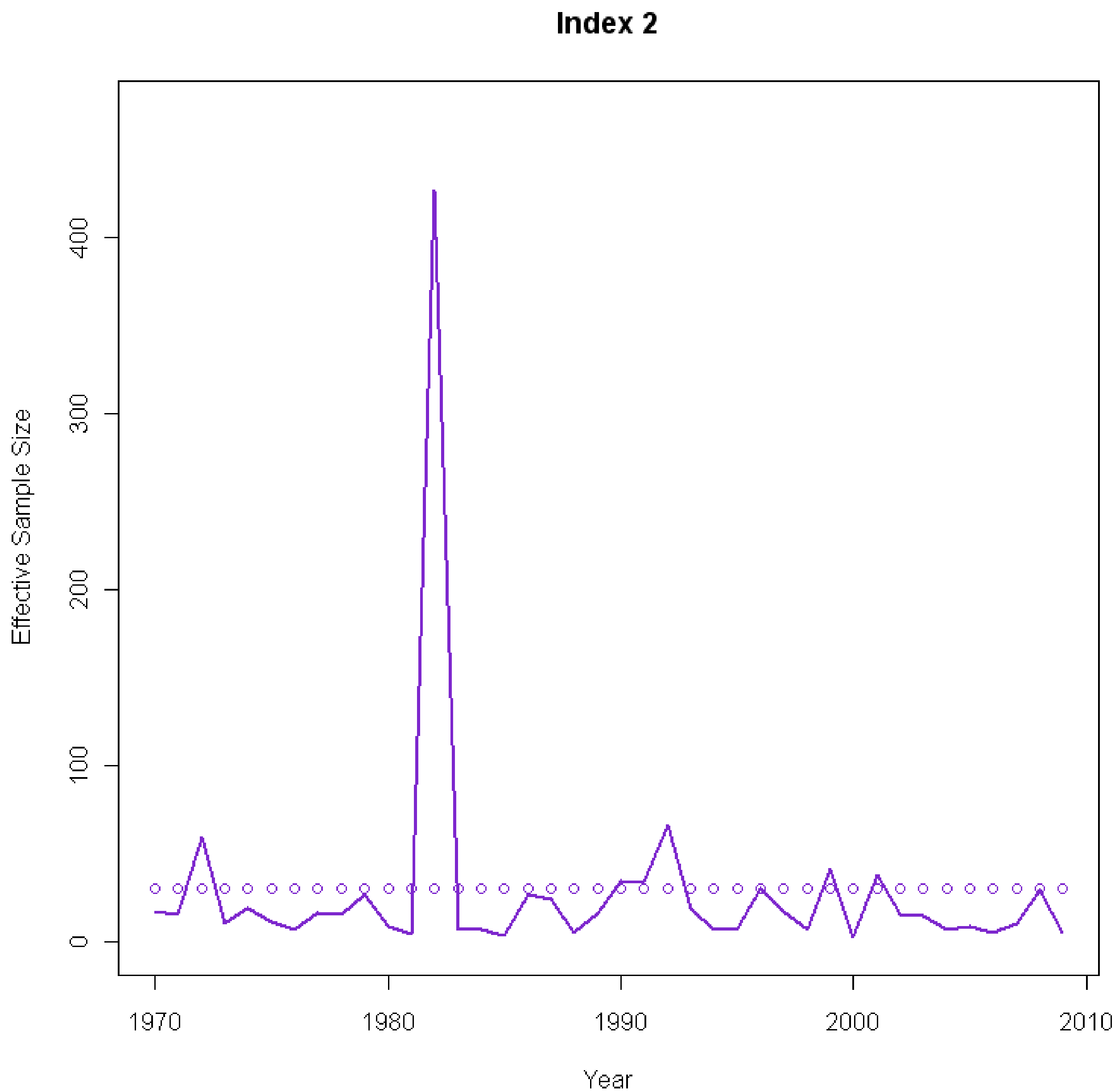


Figure C51. ASAP base model comparison of input effective sample size versus the model estimated effective sample size for the NEFSC fall index.

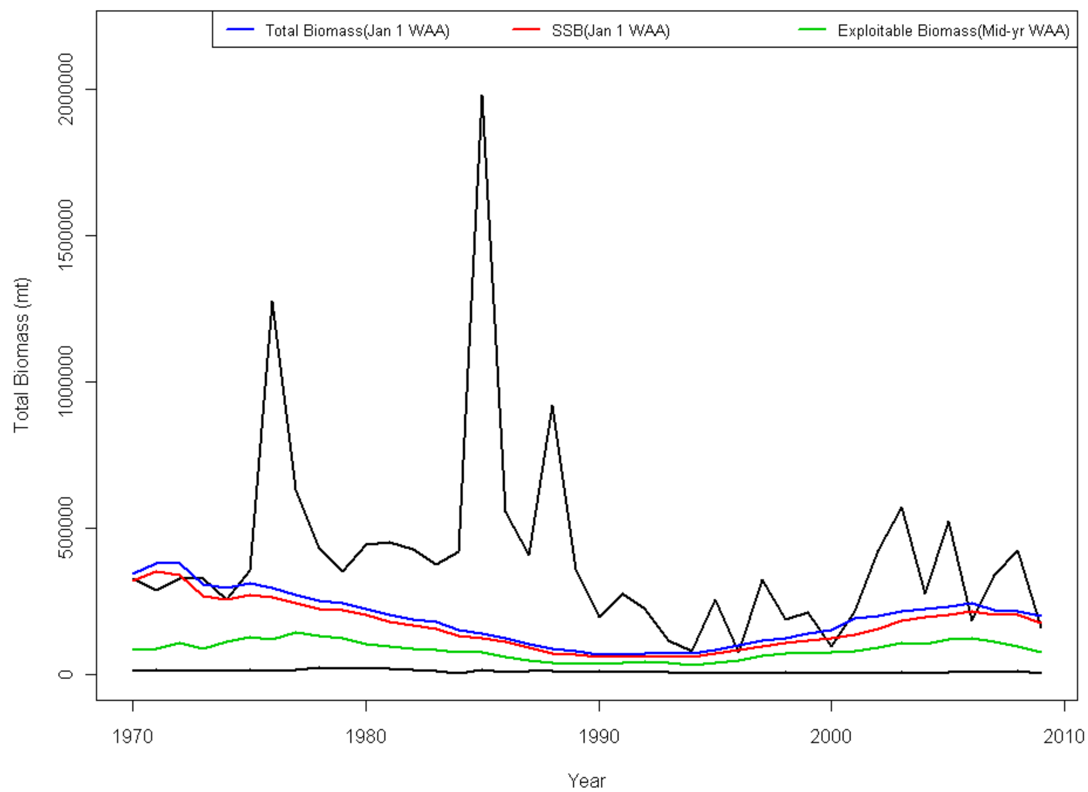


Figure C52. A proposed envelope of reasonable biomass is bounded by the solid black lines, while the ASAP base model estimated biomass of 3 quantities is plotted.

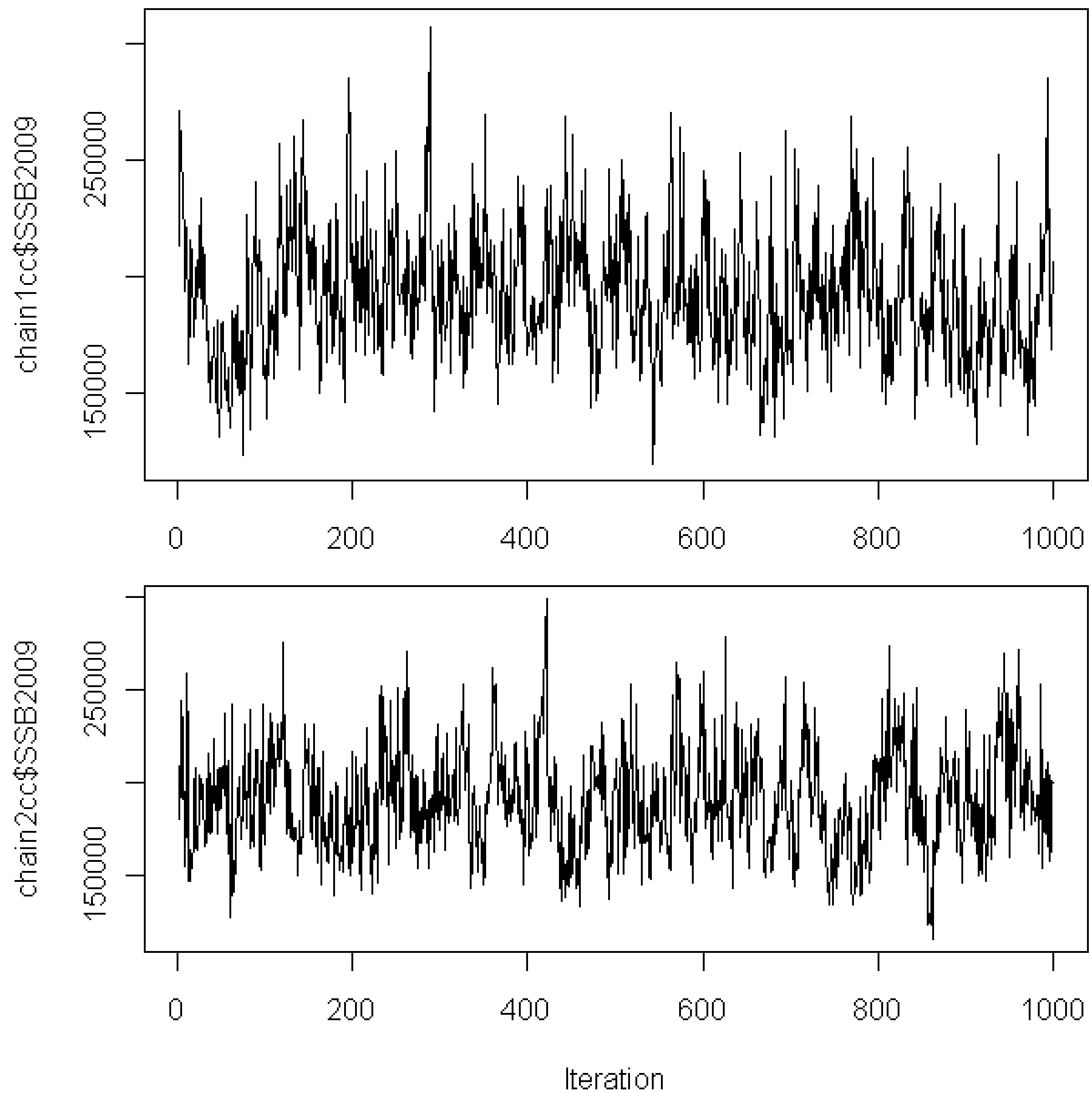


Figure C53. Trace of two MCMC chains for SSB2009, showing good mixing (ASAP base model). Each chain had initial length of 10 million; the first 5 million were dropped for burn-in, and the remaining 5 million were thinned at a rate of one out of every 5,000th. The final chain length was 1000 saved draws.

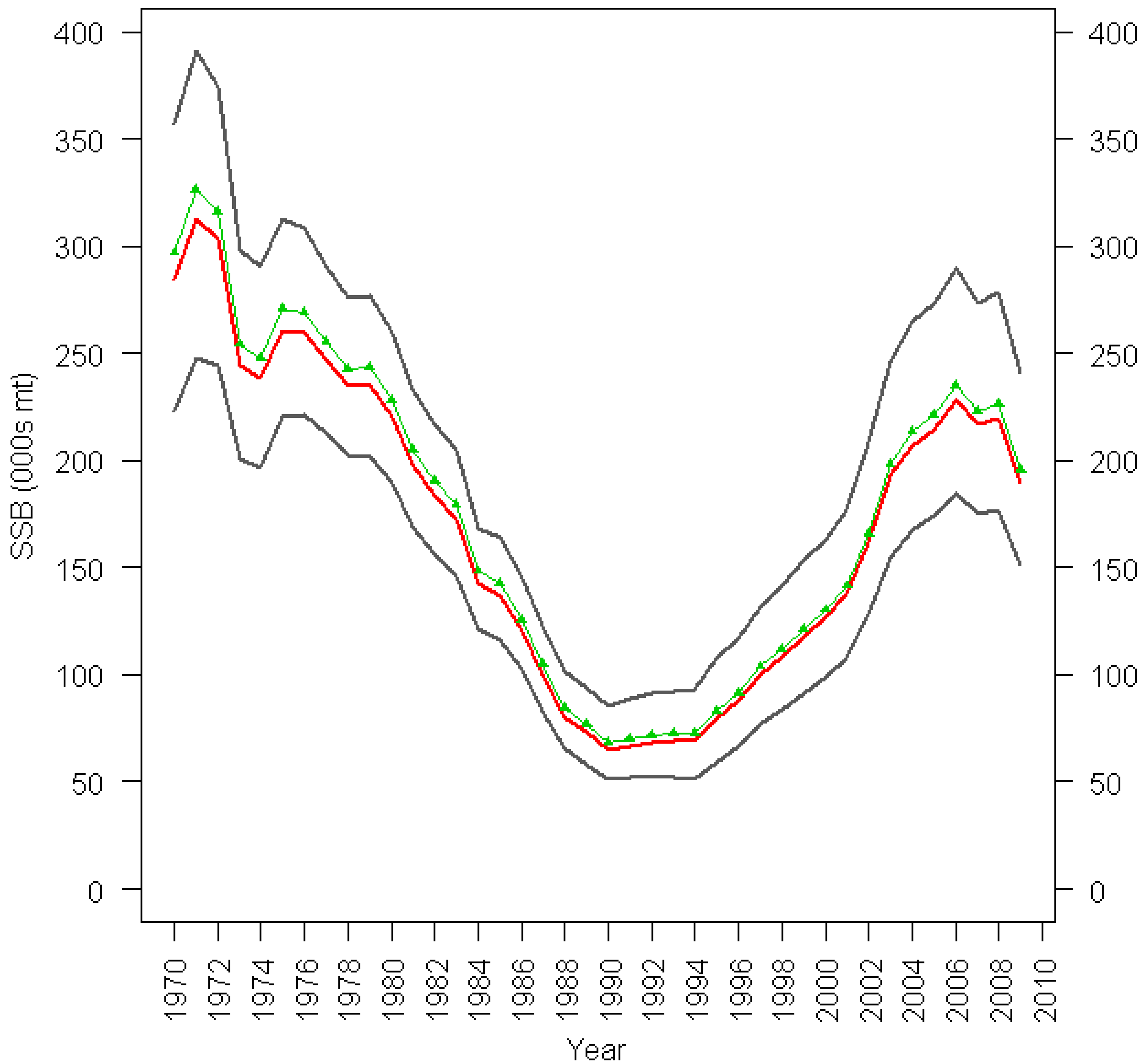


Figure C54**.** A 90% probability interval for pollock spawning stock biomass (SSB) in thousands of mt is plotted for the entire time series. The median value is in red, while the 5th and 95th percentiles are in dark grey. The point estimate from the base model (joint posterior modes) is shown in the thin green lined with filled triangles. (ASAP base model)

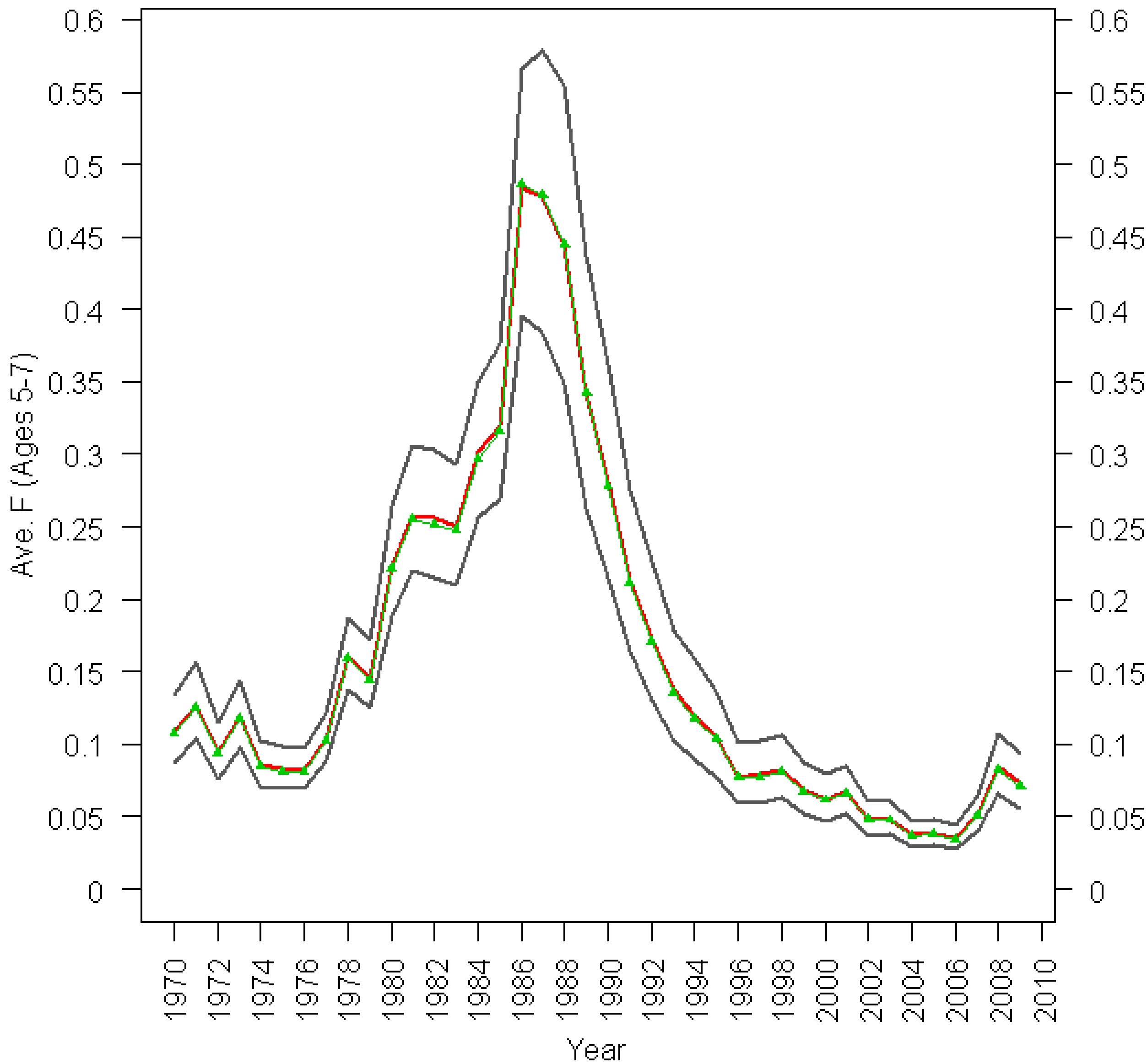


Figure C55. A 90% probability interval for the average F on ages 5-7 ($F_{5\text{-}7}$) for pollock is plotted for the entire time series. The median value is in red, while the 5th and 95th percentiles are in dark grey. The point estimate from the base model (joint posterior modes) is shown in the thin green lined with filled triangles. (ASAP base model)

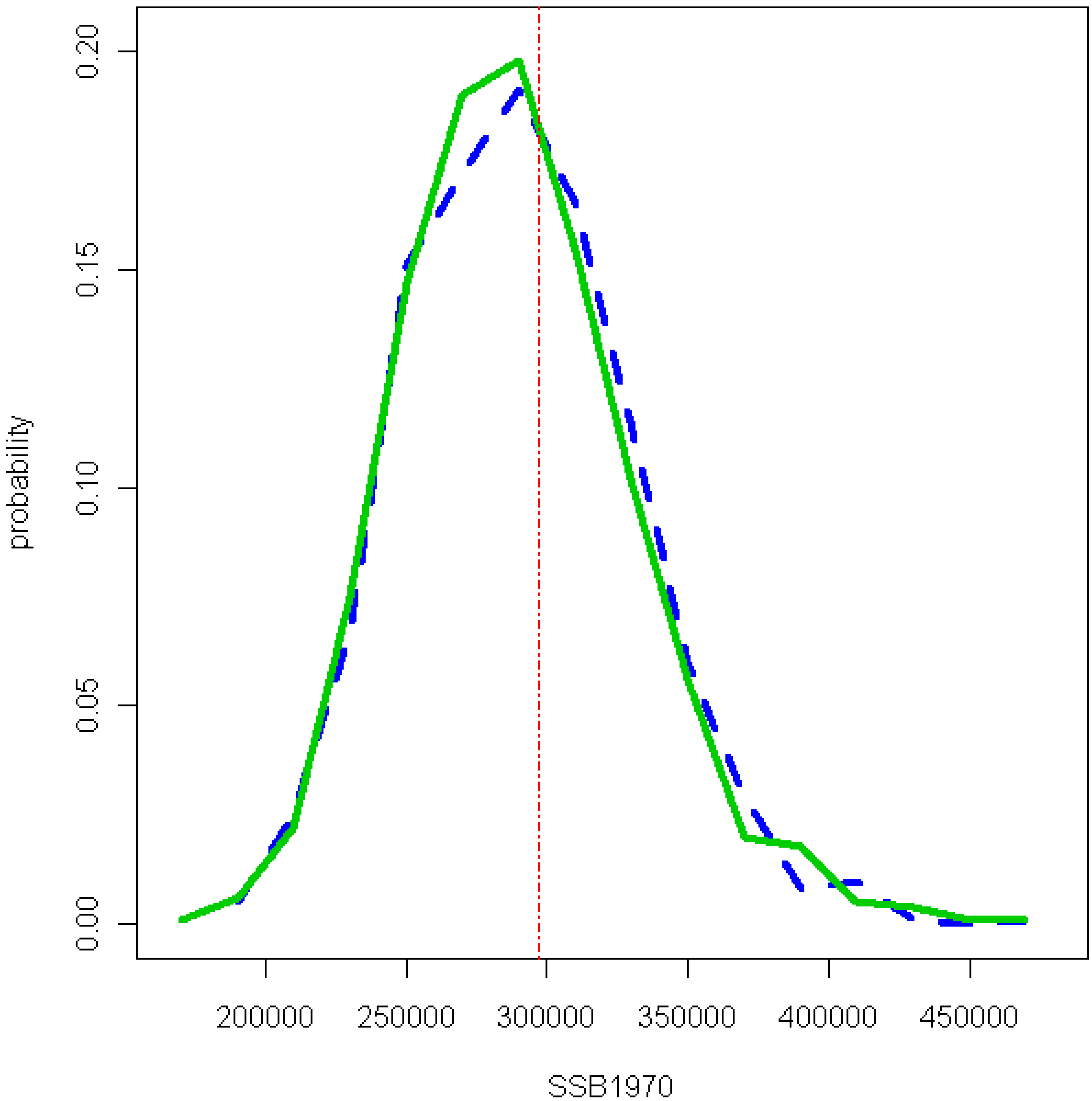


Figure C56a**.** Posterior distribution for spawning stock biomass (SSB) in 1970 (the first model year) for two MCMC chains (dotted blue and solid green lines). The vertical dashed red line indicates the point estimate. (ASAP base model)

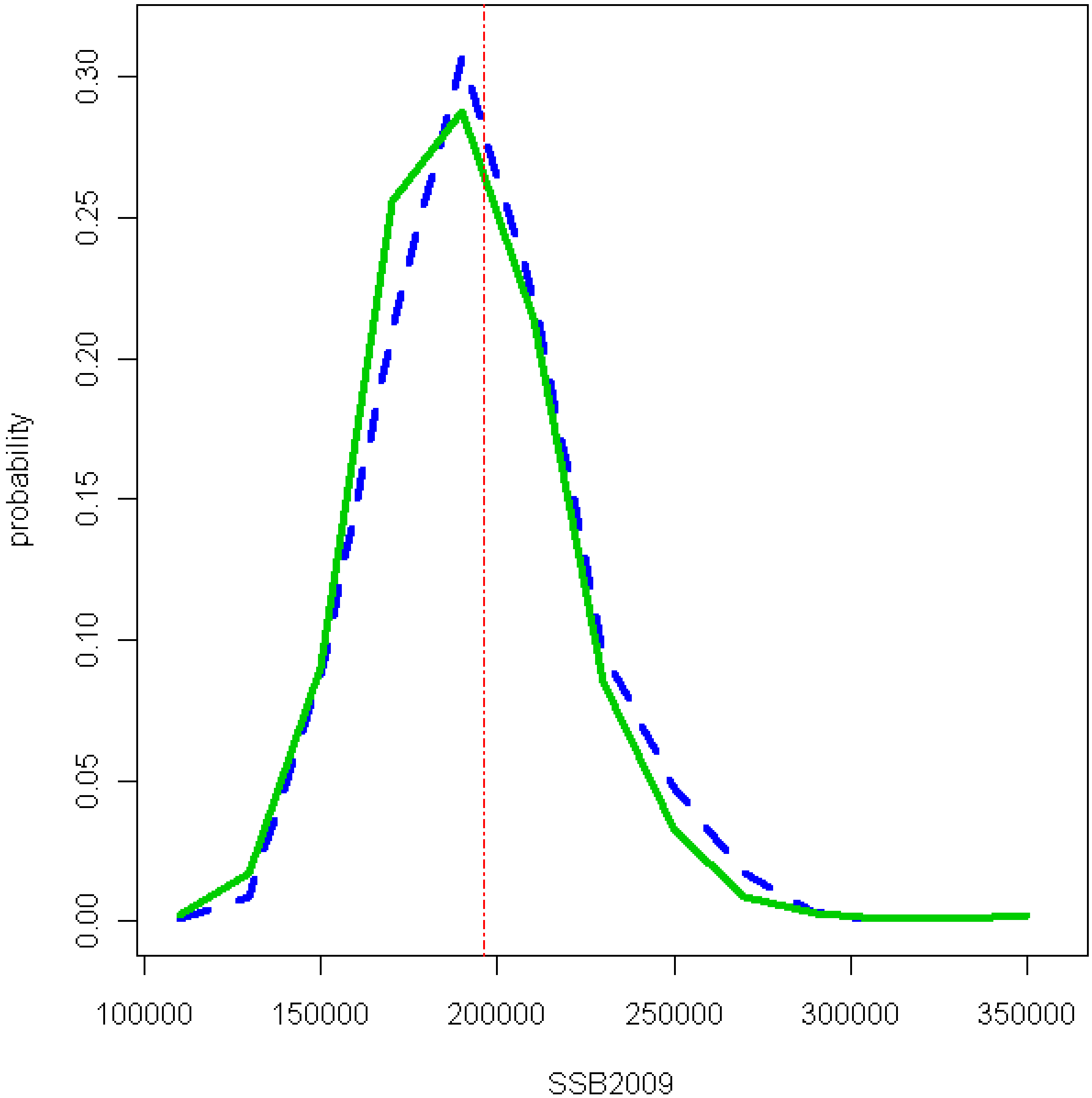


Figure C56b**.** Posterior distribution for spawning stock biomass (SSB) in 2009 for two MCMC chains (dotted blue and solid green lines). The vertical dashed red line indicates the point estimate. (ASAP base model)

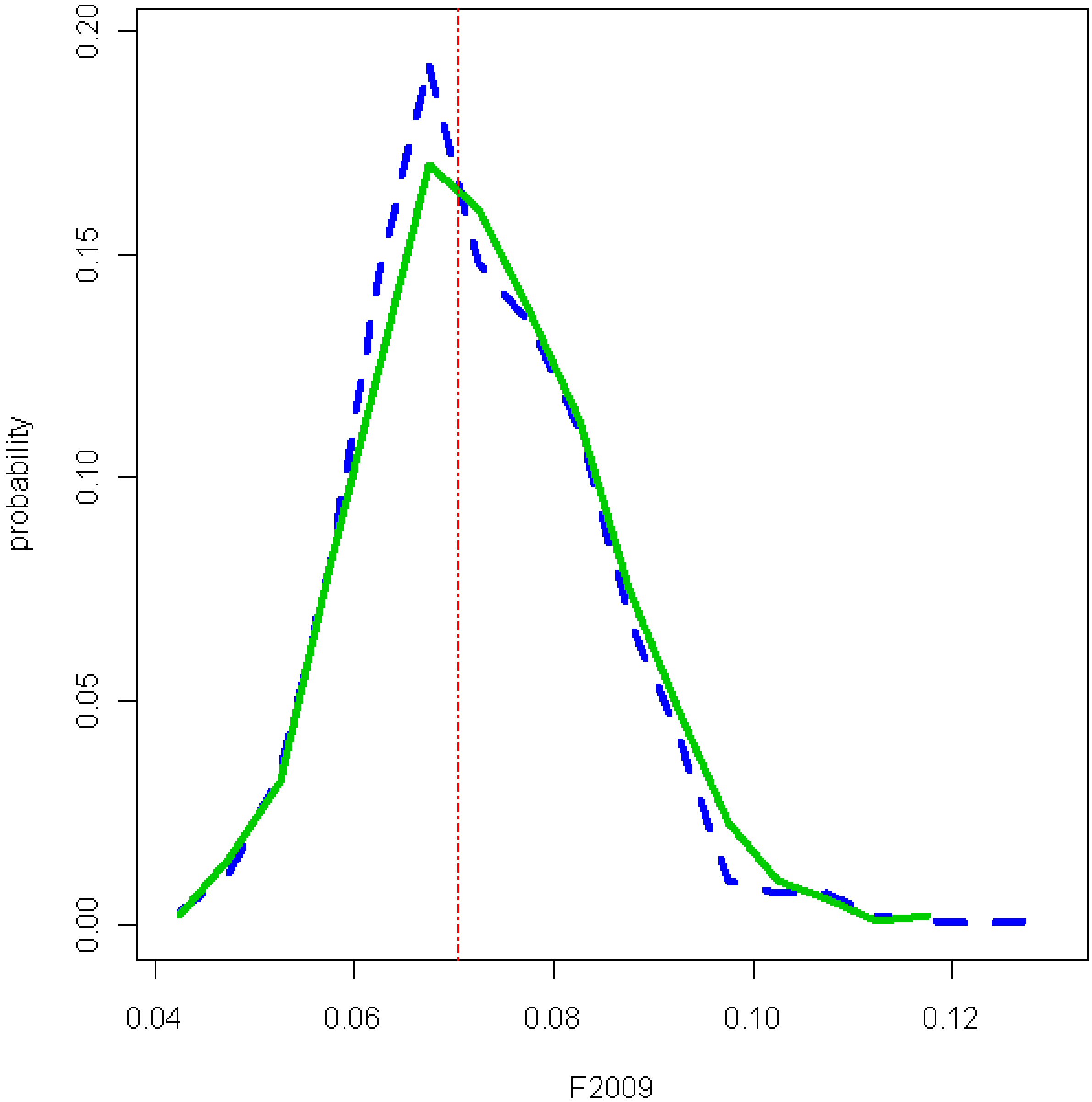


Figure C57**.** Posterior distribution for the average F on ages 5-7 ($F_{5\text{-}7}$) in 2009 for two MCMC chains (dotted blue and solid green lines). The vertical dashed red line indicates the point estimate. (ASAP base model)

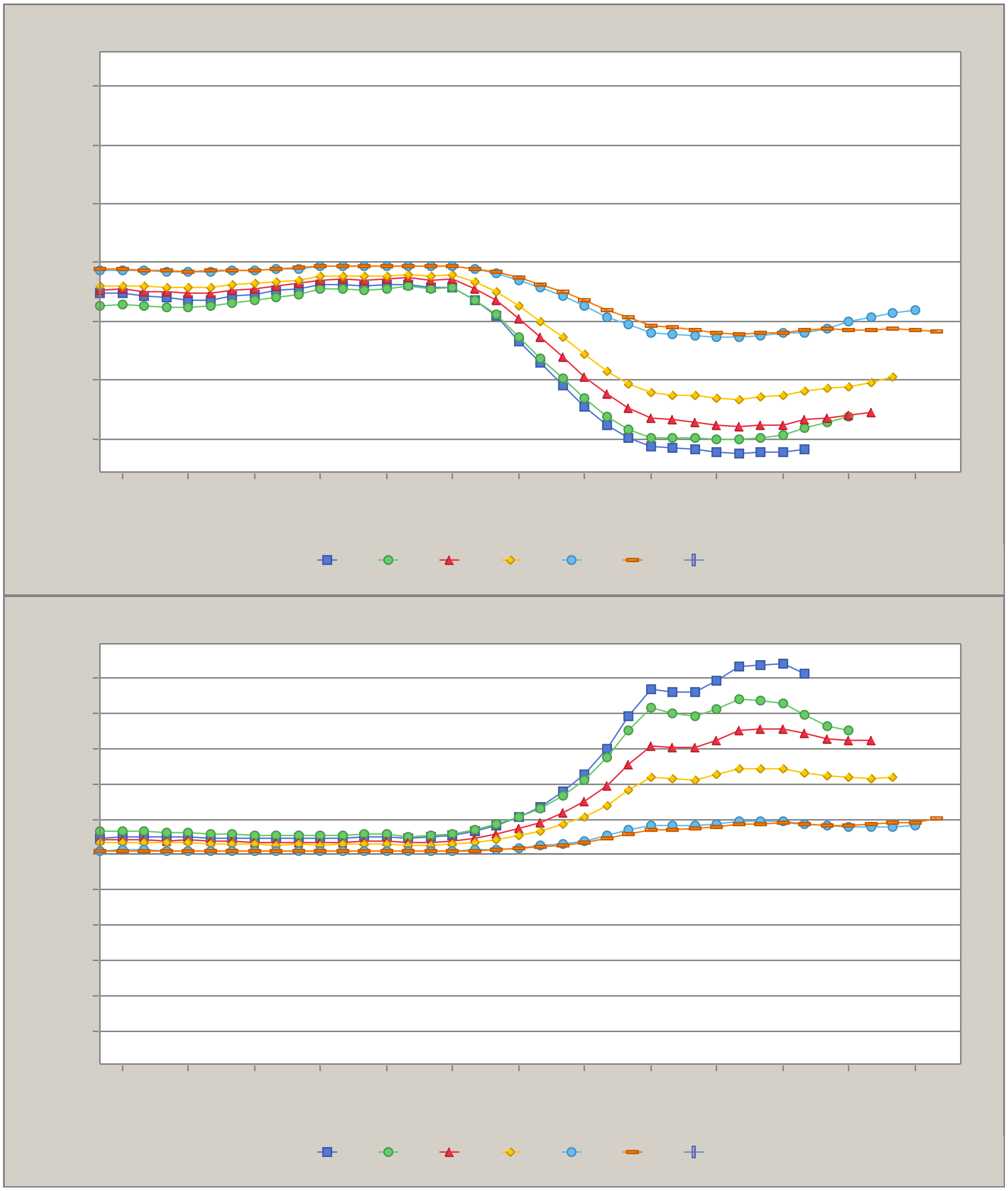


Figure C58. Retrospective analysis for years 2002-2008 for the ASAP sensitivity model with selectivity at ages 6-9+ fixed at 1.0. Relative bias for F (top) and SSB (bottom) are displayed for 2002 and 2004-2008; the model did not successfully run for year 2003.

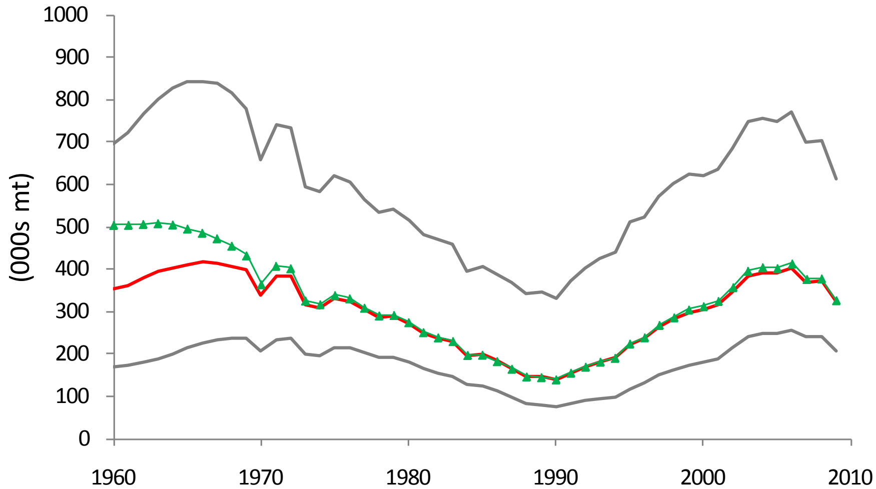


Figure C59**. A** 90% probability interval for spawning stock biomass (SSB) in thousands of mt is plotted for the entire time series. The median value is in red, while the 5th and 95th percentiles are in dark grey. The point estimate from the base model (joint posterior modes) is shown in the thin green lined with filled triangles. (model SCAA2)

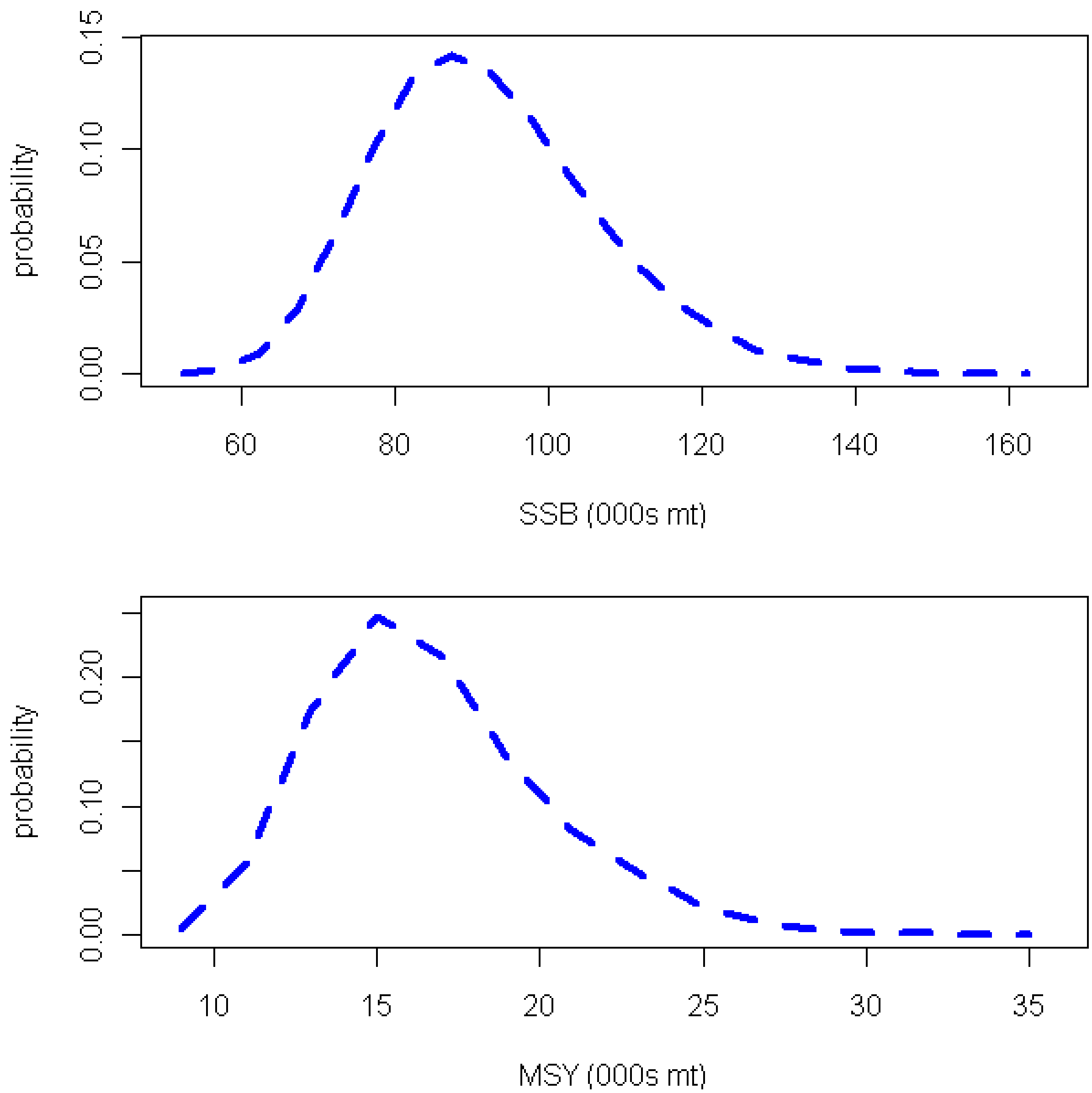


Figure C60. Distributions of SSB_{MSY} and MSY based on stochastic projections at F40%. The median estimates are 91,000 mt for SSB_{MSY} and 16,200 mt for MSY, based on projections that used F40% as a proxy for F_{MSY}. (ASAP base model)

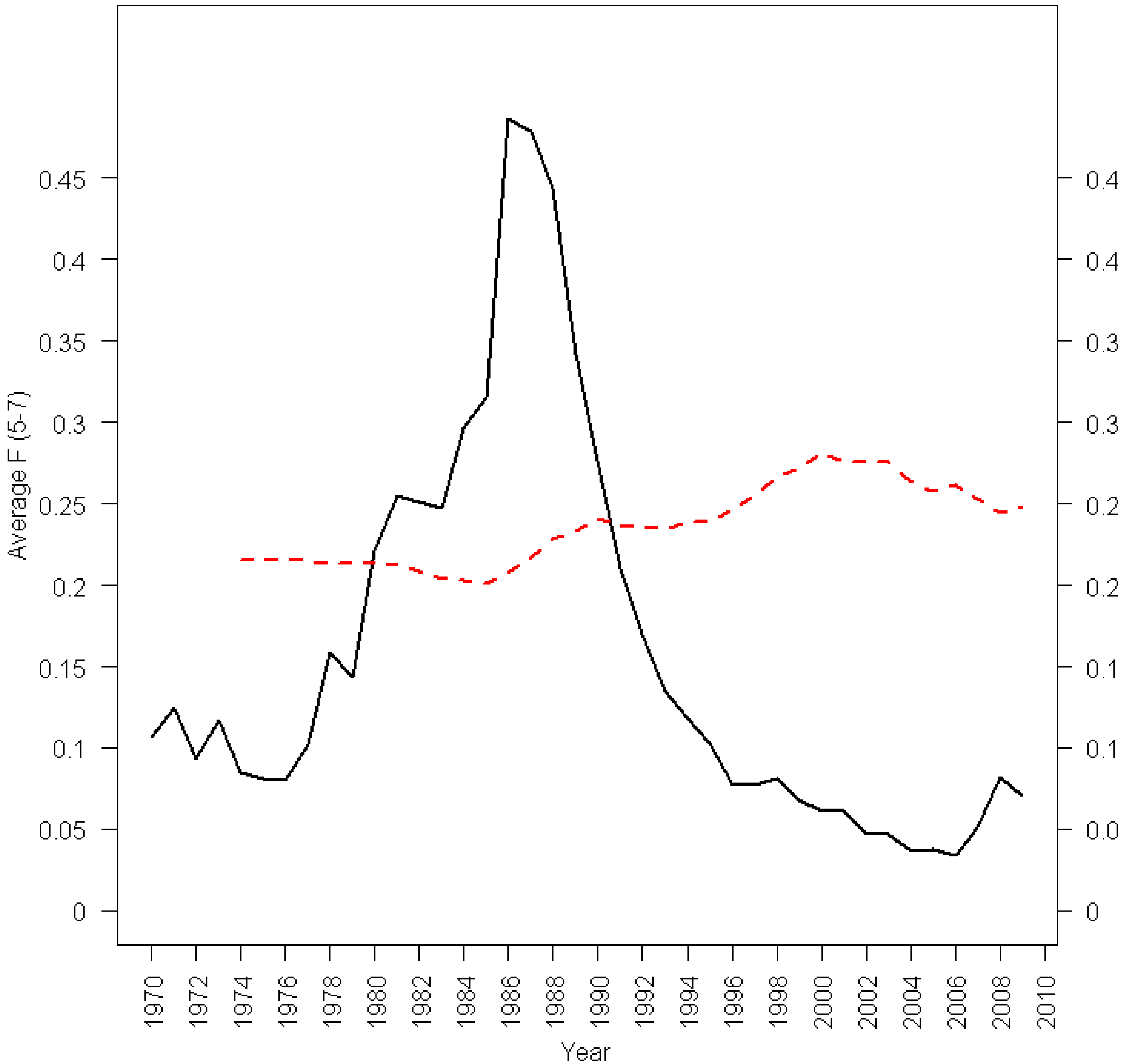


Figure C61. ASAP base model estimated time series of F_{5-7} (solid line). The dashed red line is the corresponding F40% on ages 5-7 calculated for years 1974-2009 with a 5 year moving average of weights at age, selectivity at age, and maturity at age. The F40% in 1974 used years (1970-1974) while the final F40% used years (2005-2009). (ASAP base model)

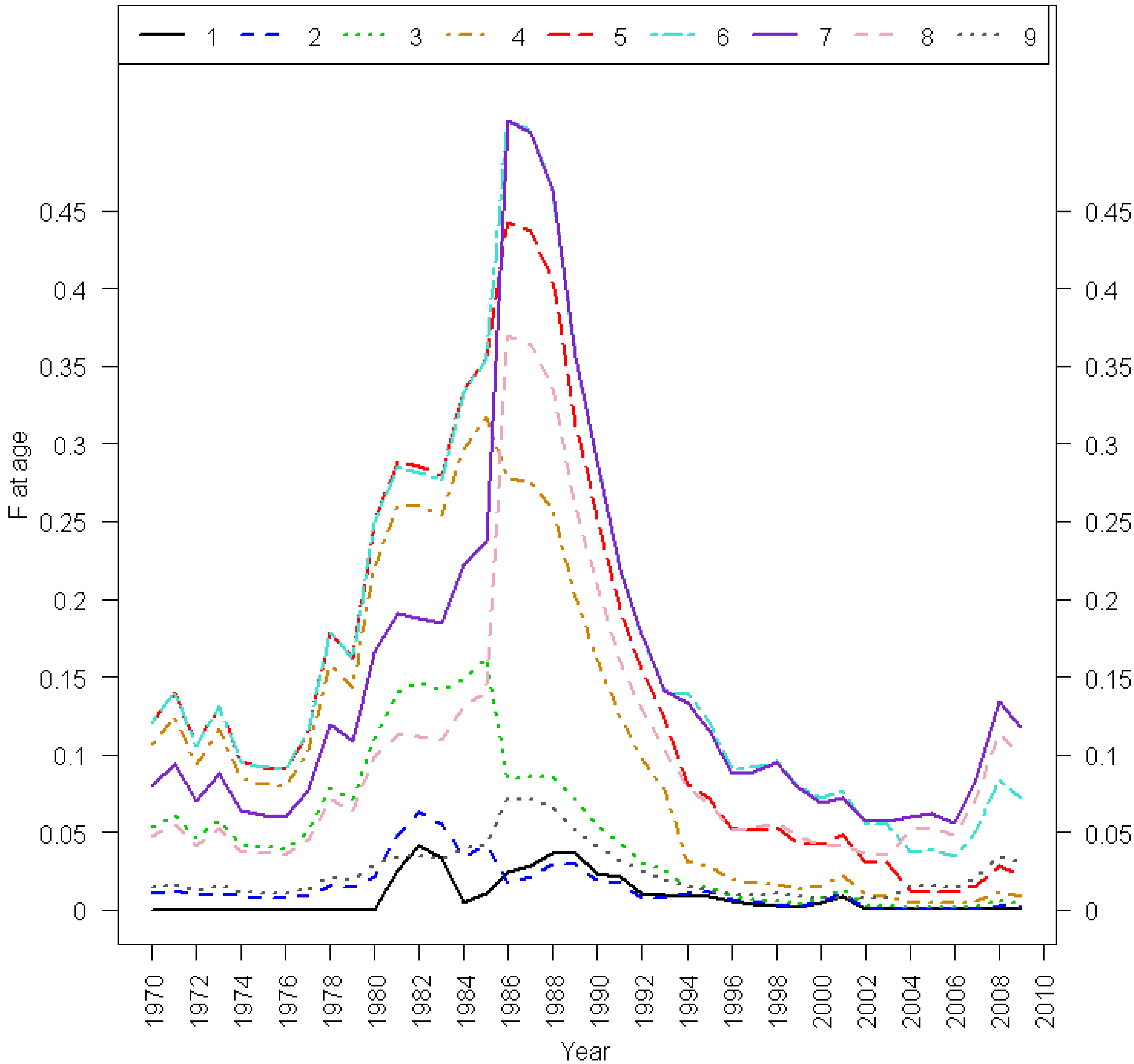


Figure C62. ASAP base model estimate of fishing mortality at age.

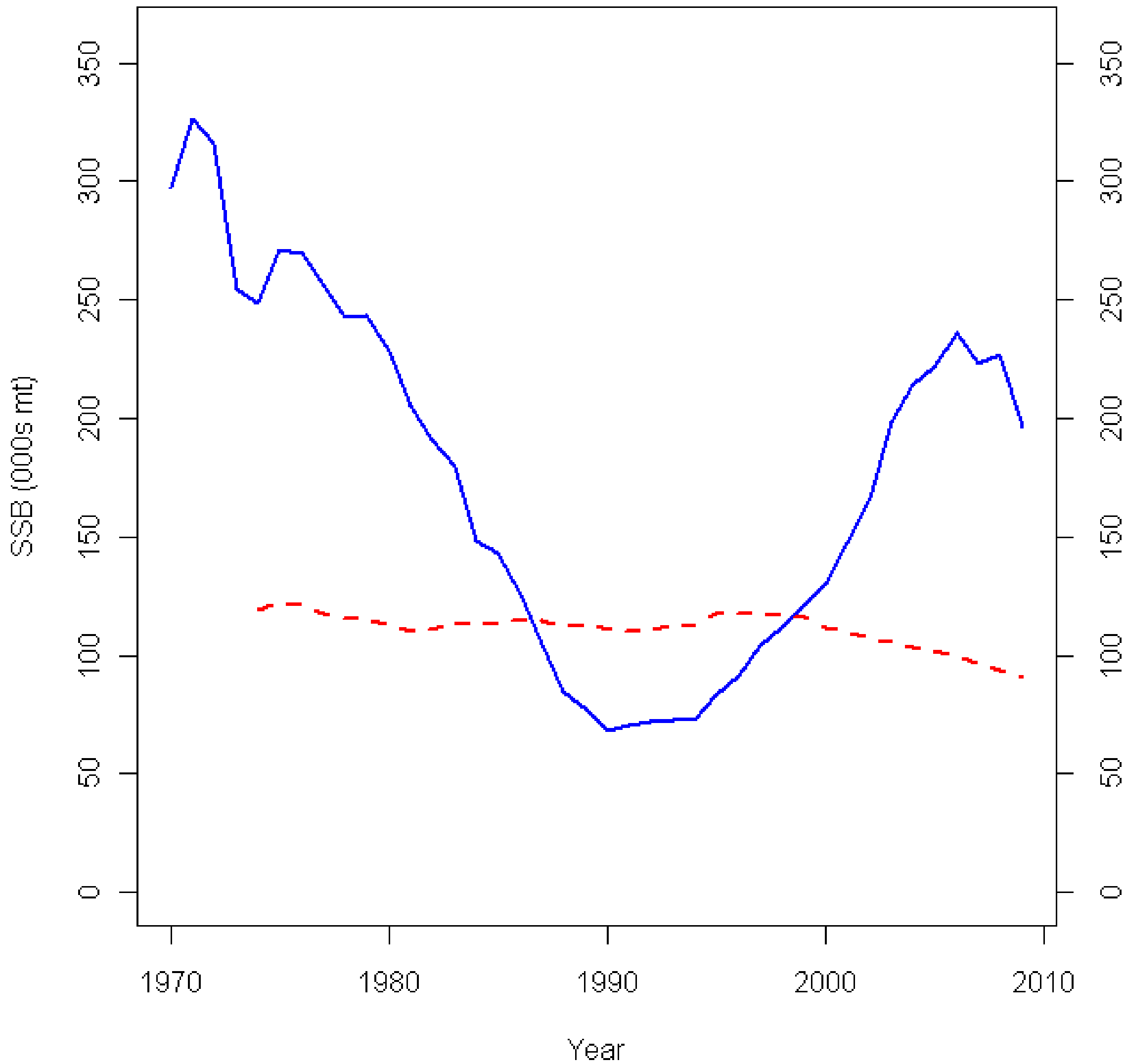


Figure C63. ASAP base model estimated time series of SSB (solid line). The dashed red line is the corresponding SSB_{MSY} proxy as calculated from stochastic projections at year-specific F40% with a 5 year moving average of weights at age, selectivity at age, and maturity at age. SSB_{MSY} in 1974 used years (1970-1974) while the final SSB_{MSY} used years (2005-2009). (ASAP base model)

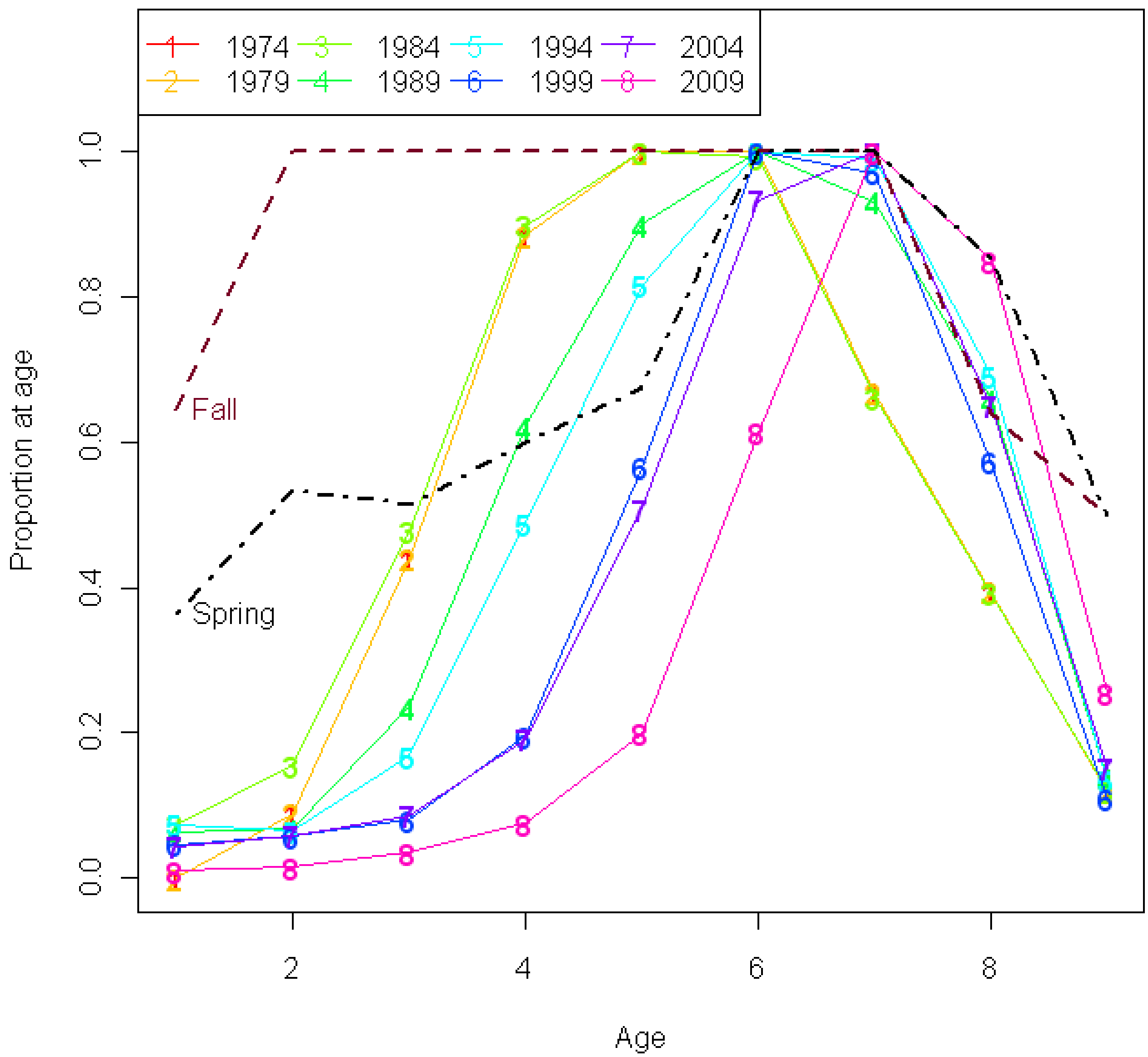


Figure C64. ASAP base model estimates for NEFSC Fall and Spring index selectivities (dashed, and dot-dash, respectively) compared to 5-year average fleet selectivities. Average selectivity at age for the 1st 5-year period includes estimates from 1970-1974 (line with '1' for point symbols) while the last 5-year average includes estimates from 2005-2009 (line with '8' for point symbols).

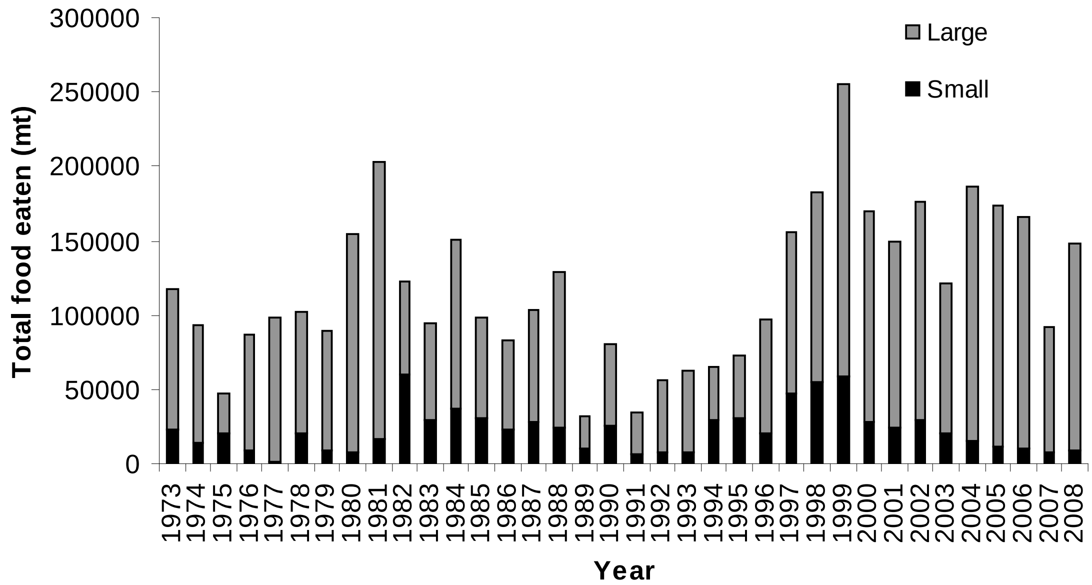


Figure C65. Total amount of food consumed by pollock.

Appendix C1: SAW50 Meeting with Pollock Fishermen

January 22 2010 – Mass DMF Annisquam River Marine Fisheries Field Station, Gloucester MA. This summary includes comments and discussions from the meeting and subsequent correspondence.

Discussion

General Approach –

Liz Brooks presented a brief review of the assessment history of pollock, plans for the benchmark assessment and some data exploration. The pollock assessment was based on a virtual population analysis from the late 1980s to the mid 1990s, but the approach was replaced with a survey index approach because of few samples in the mid 1990s. The current method of assessing and managing pollock cannot be continued, because the Albatross survey ended in 2008, and results from the calibration experiment are not expected to allow comparison of Bigelow and Albatross survey series. The general approach for the 2010 benchmark assessment is to develop an age-based model that incorporates fishery and survey data to replace the current index-based assessment method and overfishing definition.

Surveys –

The survey data currently available are the Albatross spring and fall surveys (discontinued in fall 2008, replaced with the Bigelow survey in 2009), the Gulf of Maine shrimp survey (which only surveys shrimp habitat in the western Gulf of Maine), the inshore Massachusetts survey (which samples state waters, and typically catches only small pollock). A request was made to get pollock data from the Maine-New Hampshire survey, which might provide a recruitment index similar to the Massachusetts survey. A question was also raised whether Pollock are seen on the acoustic survey, and this will be examined.

All surveys are somewhat ‘noisy’ with large inter-annual fluctuations. There was general consensus that monitoring trends in the pollock resource is difficult with trawl surveys, because of pollock behavior and distributional patterns:

- Pollock are distributed more off-bottom than other groundfish. Gillnet fishermen typically catch more pollock by adding meshes to increase the height off bottom. Catches of pollock in gillnets typically decrease when there is large dogfish bycatch, presumably because nets drop with the weight of dogfish. Off bottom behavior is particularly apparent in March and April.
- Pollock are more abundant over hard bottom, and unless surveys are designed to trawl hard-bottom, they will miss many concentrations.
- Pollock have an extremely patchy distribution. This ‘hit or miss’ aspect of pollock is shown by surveys that have many tows with no pollock and a few tows with pollock.
- Pollock are strong swimmers, with endurance to out-swim trawls.
- Availability of pollock varies seasonally. They are typically more catchable as temperatures cool in the fall. Increased catchability may be associated with spawning, more on-bottom distribution or seasonal movement patterns
- Pollock school by size, with large concentrations of fish of a similar size.
- Pollock behavior appears to have changed, with different patterns than 15 years ago.
- Inshore surveys may be too slow. Fishermen’s experience is that you have to tow at least 3 knots to catch any Pollock and the best speed is 3.5 knots.

Environmental factors that may help explain pollock availability and catchability were identified:

- Pollock is considered to be a cold-water species, and survey catches may be associated with cold temperature.
- Fishermen also observed that pollock are typically following concentrations of sand lance. Tidal stage (slack tides are favored) and moon phase might be associated with greater probability of encountering Pollock; gillnetters catch more at night (exploration of trawl survey indicated no consistent difference between catches of day and night tows)
- Catchability of pollock may also be influenced by midwater trawling, which may disrupt pollock schooling or feeding.
- Pollock get 'spooked' by gear, and move higher in the water column after a pass is made with gear; some waiting is required before Pollock are likely to re-settle towards the bottom.

One fisherman asked why the 2005 fall survey index was excluded from the stock status determination during the GARM. Although the answer wasn't clear at the meeting, correspondence after the meeting revealed that GARM III reported the status of pollock based on only one year of the trawl survey rather than a three-year centered moving average (e.g., stock size for 2000 is the average of 1999-2001), as the criteria was established by the Reference Point Working Group in 2002. When the 2008 fall trawl survey results became available a few months after the GARM, the stock was confirmed to be overfished in 2007 based on the centered three-year moving average of the trawl survey (2006-2008).

The focus of the presentation was on how the assessment can be improved using currently available data. The group requested that the benchmark assessment also identify what information would improve future assessments. Given the difficulty indexing abundance of pollock with a trawl survey, an industry-based fixed-gear survey (e.g., variable-mesh gillnet) might complement existing survey programs. Similarly, acoustic surveys might help to assess pollock and other off-bottom species that are not well sampled by bottom trawls.

Fisheries –

The series of commercial landings was reviewed. The increase in recent commercial catches was interpreted as increased availability of pollock in recent years.
Fishermen considered the pattern of landings to be largely influenced by regulations. For example peak landings in the mid-1980s were composed of much smaller fish than are retained by the large-mesh that is currently regulated. Restrictions on roller gear do not allow fishing hard bottom. Days-at-sea restrictions also did not allow exploratory fishing for concentrations of pollock or fishing in hard-bottom areas that require mending nets at sea.

Fishermen don't often target pollock, but they felt that when they do target pollock they usually can find them. The market has also held the landings lower than they could have been in recent years. Several years ago the United States government changed their criteria for pollock bids and we lost the military markets (they allow twice frozen fillets) all that market has moved to the west coast pollock. Before that pollock was worth $0 .70 to $1.00 per pound on a consistent basis. Since then, pollock value can be as low as $0.35 cents. Therefore, many boats have not targeted pollock due to relatively low cost fish price, high labor costs to dress and higher fuel costs. Traditional fishing grounds are currently closed to commercial fishing. For example concentrations of pollock are in the western Gulf of Maine closure, just east of 70o 15'W. Traditional fishing grounds were also in the Cashes Ledge closure.

Many pollock were also traditionally caught Down East and into the Bay of Fundy. Vessels no longer fish there because it is too far to go for cheap fish and high fuel costs, and the Hague Line was established. On George's Bank the larger boats fishing east of the Hague Line used to catch very large quantities of pollock this traditional fishing ground is no longer available to US fishermen.

The apparent increase in recreational landings (e.g., a substantial increase in 2008) was considered to be realistic. The increase was considered to result from concentrations of pollock in areas that are closed to commercial fishing, and a general increase in availability of pollock in recent years. It was suggested that recreational catch included small fish, despite the recreational size limit. This information is considered anecdotal at present, until size samples can be examined.

Participation in the meeting and candid contributions were appreciated. The meeting was informative for all participants, and the information presented at the meeting will be considered in the development of the benchmark assessment. Participation in the upcoming data meeting, model meeting and SARC were also encouraged.

Appendix C2: Statistical Catch-at-Age Analysis Methodology

The model equations and the general specifications of the SCAA methodology applied are described below, followed by details of the contributions to the (penalised) log-likelihood function from the different sources of data available and assumptions concerning the stock-recruitment relationship. Quasi-Newton minimization is used to minimize the total negative log-likelihood function (the package AD Model Builder™, Otter Research, Ltd is used for this purpose).

B1. Population dynamics

B1.1 Numbers-at-age

The resource dynamics are modelled by the following set of population dynamics equations:

$$N_{y+1,1} = R_{y+1} \qquad \text{(B1)}$$

$$N_{y+1,a+1} = \left(N_{y,a}\, e^{-M_a/2} - \sum_f C^f_{y,a} \right) e^{-M_a/2} \qquad \text{for } 1 \le a \le m-2 \qquad \text{(B2)}$$

$$N_{y+1,m} = \left(N_{y,m-1}\, e^{-M_{m-1}/2} - \sum_f C^f_{y,m-1} \right) e^{-M_{m-1}/2} + \left(N_{y,m}\, e^{-M_m/2} - \sum_f C^f_{y,m} \right) e^{-M_m/2} \qquad \text{(B3)}$$

where

$N_{y,a}$ is the number of fish of age *a* at the start of year *y* (which refers to a calendar year),

R_y is the recruitment (number of 1-year-old fish) at the start of year *y*,

M_a denotes the natural mortality rate for fish of age *a*,

$C^f_{y,a}$ is the predicted number of fish of age *a* caught in year *y* by fleet *f*, and

m is the maximum age considered (taken to be a plus-group).

B1.2. Recruitment

The number of recruits (1-year olds) at the start of year *y* is assumed to be related to the spawning stock size (i.e. the biomass of mature fish) by a Beverton-Holt or a modified (generalised) form of the Ricker stock-recruitment relationship, parameterised in terms of the "steepness" of the stock-recruitment relationship, *h*, and the pre-exploitation equilibrium spawning biomass, SSB_0, and recruitment, R_0 and allowing for annual fluctuation about the deterministic relationship:

$$R_{y+1} = \frac{4hR_0 SSB_y}{SSB_0(1-h) + (5h-1)SSB_y} e^{(\varsigma_y - \sigma_R^2/2)} \qquad \text{(B4)}$$

for the Beverton-Holt stock-recruitment relationship and

$$R_{y+1} = \alpha SSB_y \exp\left(-\beta (SSB_y)^\gamma\right) e^{(\varsigma_y - \sigma_R^2/2)} \qquad \text{(B5)}$$

with

$$\alpha = R_0 \exp\left(\beta (SSB_0)^\gamma\right) \qquad \text{and} \qquad \beta = \frac{\ln(5h)}{(SSB_0)^\gamma (1 - 5^{-\gamma})}$$

for the modified Ricker relationship (for the true Ricker, =1)

where

ς_y reflects fluctuations about the expected recruitment for year *y*, which are assumed to be normally distributed with standard deviation σ_R (which is input in the applications considered here); these residuals are treated as estimable parameters in the model fitting process.

SSB_y is the spawning biomass at the start of year *y*, computed as:

$$SSB_y = \sum_{a=1}^{m} f_{y,a} w_{y,a}^{strt} N_{y,a} \tag{B6}$$

where

$w_{y,a}^{strt}$ is the mass of fish of age *a* at the beginning of the year (Table A6), and

$f_{y,a}$ is the proportion of fish of age *a* that are mature (Table A5).

In the fitting procedure, SSB_0 is estimated while *h* can be estimated or fixed. For the Beverton-Holt form, *h* is bounded above by 0.9 to preclude high recruitment at extremely low spawning biomass, whereas for the modified Ricker form, *h* is bounded above by 1.5 to preclude extreme compensatory behaviour.

B1.3. Total catch and catches-at-age

The fleet-disaggregated catch by mass in year *y* is given by:

$$C_y^f = \sum_{a=1}^{m} w_{y,a}^{f,mid}\, C_{y,a}^f = \sum_{a=1}^{m} w_{y,a}^{f,mid}\, N_{y,a}\, e^{-M_a/2}\, S_{y,a}^f\, F_y^f \tag{B7}$$

where

$w_{y,a}^{f,mid}$ denotes the mass of fish of age *a* landed in year *y* (Tables A7, A8 and A9),

$C_{y,a}^f$ is the catch-at-age, i.e. the number of fish of age *a*, caught in year *y* by fleet *f*,

$S_{y,a}^f$ is the commercial selectivity of fleet *f* (i.e. combination of availability and vulnerability to fishing gear) at age *a* for year *y*; when $S_{y,a}$ = 1, the age-class *a* is said to be fully selected, and

F_y^f is the proportion of a fully selected age class that is fished, for fleet *f*.

B1.4. Initial conditions

For the first year (y_0) considered in the model, the stock is assumed to be at a fraction (θ) of its pre-exploitation biomass, i.e.:

$$SSB_{y_0} = \theta \cdot SSB_0 \tag{B8}$$

with the starting age structure:

$$N_{y_0,a} = R_{start} N_{start,a} \qquad \text{for } 1 \le a \le m \tag{B9}$$

where

$$N_{start,1} = 1 \tag{B10}$$

$$N_{start,a} = N_{start,a-1} e^{-M_{a-1}} (1 - \phi S_{a-1}) \qquad \text{for } 2 \le a \le m-1 \tag{B11}$$

$$N_{start,m} = N_{start,m-1} e^{-M_{m-1}} (1 - \phi S_{m-1}) \big/ (1 - e^{-M_m} (1 - \phi S_m)) \tag{B12}$$

where ϕ characterises the average fishing proportion over the years immediately preceding y_0.

B2. The (penalised) likelihood function

The model can be fit to survey indices and catch-at-age as well as commercial catch-at-age data to estimate model parameters (which may include residuals about the stock-recruitment function, through the incorporation of a penalty function described below). Contributions by each of these to the negative of the (penalised) log-likelihood ($-\ell nL$) are as follows.

B2.1 Survey relative abundance data

The likelihood is calculated assuming that an observed index for a particular survey is log-normally distributed about its expected value:

$$I_y^i = \hat{I}_y^i \exp\left(\varepsilon_y^i\right) \quad \text{or} \quad \varepsilon_y^i = \ell n\left(I_y^i\right) - \ell n\left(\hat{I}_y^i\right) \tag{B13}$$

where

I_y^i is the survey index for year *y* and series *i*,

$\hat{I}_y^i = \hat{q}^i \hat{B}_y^{surv}$ is the corresponding model estimate, where

$$\hat{B}_y^{surv} = \sum_{a=1}^{m} S_a^{surv} N_{y,a} e^{-\frac{M_a}{4}} \left(1 - \sum_f S_{y,a}^f F_y^f \Big/ 4\right) \tag{B14}$$

for spring surveys,

$$\hat{B}_y^{surv} = \sum_{a=1}^{m} S_a^{surv} N_{y,a} e^{-\frac{M_a}{2}} \left(1 - \sum_f S_{y,a}^f F_y^f \Big/ 2\right) \tag{B15}$$

for summer surveys,

$$\hat{B}_y^{surv} = \sum_{a=1}^{m} S_a^{surv} N_{y,a} e^{-\frac{3M_a}{4}} \left(1 - 3\sum_f S_{y,a}^f F_y^f \Big/ 4\right) \tag{B16}$$

for fall surveys,

$$\hat{B}_y^{surv} = B_y^{sp} \tag{B17}$$

for the larval index, and

$\hat{q}^i$ is the constant of proportionality (catchability) for survey series *i*, and

ε_y^i from $N\left(0, \left(\sigma_y^i\right)^2\right)$.

The contribution of the survey indices to the negative of the log-likelihood function (after removal of constants) is then given by:

$$-\ell n L^{surv} = \sum_i \sum_y \left[\ell n\left(\sigma_y^i\right) + \left(\varepsilon_y^i\right)^2 / 2\left(\sigma_y^i\right)^2\right] \tag{B18}$$

where

σ_y^i is the standard deviation of the residuals for the logarithm of index *i* in year *y*, taken to be given by the survey CV.

The estimated CVs likely fail to include all sources of variability, and unrealistically high precision could hence be accorded to these indices. The procedure adopted takes account of an additional variance $\left(\sigma_A^i\right)^2$ which is treated as another estimable parameter in the minimisation process, and included by replacing σ_y^i by $\sqrt{\left(\sigma_y^i\right)^2 + \left(\sigma_A^i\right)^2}$ in equation B18. This procedure is carried out enforcing the constraint that $0 \le \left(\sigma_A^i\right)^2 \le 2$.

The catchability coefficient q^i for survey index *i* is estimated by its maximum likelihood value:

$$\ell n\,\hat{q}^i = \frac{\sum_y \left(\ln I_y^i - \ln \hat{B}_y^{surv}\right) \Big/ \left(\left(\sigma_y^i\right)^2 + \left(\sigma_A^i\right)^2\right)}{\sum_y 1 \Big/ \left(\sigma_y^i\right)^2 + \left(\sigma_A^i\right)^2} \qquad \text{(B19)}$$

B2.3.Commercial catches-at-age

The contribution of the catch-at-age data to the negative of the log-likelihood function under the assumption of an “adjusted” lognormal error distribution is given by:

$$-\ell n L^{CAA} = \sum_f w_{CAA} \sum_y \sum_a \left[\ell n\left(\sigma_{com}^f / \sqrt{p_{y,a}^f}\right) + p_{y,a}^f \left(\ell n\, p_{y,a}^f - \ell n\, \hat{p}_{y,a}^f \right)^2 / 2\left(\sigma_{com}^f\right)^2 \right] \qquad \text{(B20)}$$

where

$p_{y,a}^f = C_{y,a}^f / \sum_{a'} C_{y,a'}^f$ is the observed proportion of fish caught in year *y* by fleet *f* that are of age *a*,

$\hat{p}_{y,a}^f = \hat{C}_{y,a}^f / \sum_{a'} \hat{C}_{y,a'}^f$ is the model-predicted proportion of fish caught in year *y* by fleet *f* that are of age *a*,

where

$$\hat{C}_{y,a}^f = N_{y,a}\, e^{-M_a/2}\, S_{y,a}^f\, F_y^f \qquad \text{(B21)}$$

and

σ_{com}^f is the standard deviation associated with the catch-at-age data of fleet *f*, which is estimated in the fitting procedure by:

$$\sigma_{com}^f = \sqrt{\sum_y \sum_a p_{y,a}^f \left(\ell n\, p_{y,a}^f - \ell n\, \hat{p}_{y,a}^f \right)^2 / \sum_y \sum_a 1} \qquad \text{(B22)}$$

w_{CAA} is input (this allows for the contribution from these data to be up-or downweighted compared to that from the survey indices).

The log-normal error distribution underlying equation (B20) is chosen on the grounds that (assuming no ageing error) variability is likely dominated by a combination of interannual variation in the distribution of fishing effort, and fluctuations (partly as a consequence of such variations) in selectivity-at-age, which suggests that the assumption of a constant coefficient of variation is appropriate. However, for ages poorly represented in the sample, sampling variability considerations must at some stage start to dominate the variance. To take this into account in a simple manner, motivated by binomial distribution properties, the observed proportions are used for weighting so that undue importance is not attached to data based upon a few samples only. Commercial catches-at-age are incorporated in the likelihood function using equation (B20), for which the summation over age *a* is taken from age a_{minus} (considered as a minus group) to a_{plus} (a plus group).

B2.4.Survey catches-at-age

The survey catches-at-age are incorporated into the negative log-likelihood in an analogous manner to the commercial catches-at-age, assuming an adjusted log-normal error distribution (equation B20) where:

$p_{y,a}^{surv} = C_{y,a}^{surv} / \sum_{a'} C_{y,a'}^{surv}$ is the observed proportion of fish of age a from survey *surv* in year *y*,

$\hat{p}_{y,a}^{surv}$ is the expected proportion of fish of age *a* in year *y* in the survey *surv*, given by:

$$\hat{p}^{surv}_{y,a} = \frac{S^{surv}_{a} N_{y,a} e^{-\frac{M_a}{4}} \left(1 - \sum_f S^f_{y,a} F^f_y \Big/ 4\right)}{\sum_{a'} S^{surv}_{a'} N_{y,a'} e^{-\frac{M_{a'}}{4}} \left(1 - \sum_f S^f_{y,a'} F^f_y \Big/ 4\right)} \tag{B23}$$

for spring surveys, and

$$\hat{p}^{surv}_{y,a} = \frac{S^{surv}_{a} N_{y,a} e^{-\frac{3M_a}{4}} \left(1 - 3\sum_f S^f_{y,a} F^f_y \Big/ 4\right)}{\sum_{a'} S^{surv}_{a'} N_{y,a'} e^{-\frac{3M_{a'}}{4}} \left(1 - 3\sum_f S^f_{y,a'} F^f_y \Big/ 4\right)} \tag{B24}$$

for fall surveys.

B2.5. Survey catches-at-length

The predicted proportions-at-age from equations B23 and B24, or similar equations for other surveys, may be converted into proportions-at-length using the von Bertalanffy growth equation, assuming that the length-at-age distribution remains constant over time:

$$\hat{p}^{surv}_{y,l} = \sum_a \hat{p}^{surv}_{y,a} A^{surv}_{a,l} \tag{B25}$$

where

$A^{surv}_{a,l}$ is the proportion of fish of age *a* that fall in the length group *l* for survey *surv* (i.e. $\sum_l A^{surv}_{a,l} = 1$ for all ages *a* for survey *surv*).

The matrix *A* is calculated under the assumption that length-at-age is normally distributed about a mean given by the von Bertalanffy equation, i.e.:

$$L_a \sim N\left[L_\infty \left(1 - e^{-\kappa(a-t_0)}\right); \theta^2_a\right] \tag{B26}$$

where

N is the normal distribution, and

θ_a is the standard deviation of length-at-age *a*, which is modelled to be proportional to the expected length at age *a*, i.e.:

$$\theta_a = \beta L_\infty \left(1 - e^{-\kappa(a-t_0)}\right) \tag{B27}$$

where   can be fixed or estimated in the model fitting process.

The following term is then added to the negative log-likelihood:

$$-\ \text{n}\, L^{\text{CAL}} = \sum_{surv} w_{CAL} \sum_y \sum_l \left[\ n\left(\sigma^{surv}_{len} / \sqrt{\hat{p}^{surv}_{y,l}}\right) + \hat{p}^{surv}_{y,l} \left(\ np^{surv}_{y,l} -\ n\ \hat{p}^{surv}_{y,l}\right)^2 / 2\left(\sigma^{surv}_{len}\right)^2\right] \tag{B28}$$

where

$p^{surv}_{y,l}$ is the observed proportion (by number) in length group *l* in the catch in year *y* for survey *surv*, and

σ^{surv}_{len} is the standard deviation associated with the length-at-age data for survey *surv*, which is estimated in the fitting procedure by:

$$\hat{\sigma}^{surv}_{len} = \sqrt{\sum_y \sum_l \hat{p}^{surv}_{y,l} \left(\ln p^{surv}_{y,l} - \ln \hat{p}^{surv}_{y,l}\right)^2 / \sum_y \sum_l 1} \tag{B29}$$

The w_{CAL} weighting factor may be set at a value less than 1 to downweight the contribution of the catch-at-length data to the overall negative log-likelihood compared to that of the survey or catch-at-age data. The reason that this factor is introduced is that the $p_{y,l}^{surv}$ data for a given year frequently show evidence of strong positive correlation, and so are not as informative as the independence assumption underlying the form of equation B28 would otherwise suggest.

B2.6. Stock-recruitment function residuals

The stock-recruitment residuals are assumed to be log-normally distributed. Thus, the contribution of the recruitment residuals to the negative of the (now penalised) log-likelihood function is given by:

$$-\ \ nL^{SRpen} = \sum_{y=y1}^{y2}\left[\varepsilon_y^2 \big/ 2\sigma_R^2\right] \qquad \text{(B30)}$$

where

ε_y from $N\left(0,(\sigma_R)^2\right)$, which is estimated for year *y*1 to *y*2 (see equation (B4)), and

σ_R is the standard deviation of the log-residuals, which is input (a value of 0.4 is used for the Base Case assessment).

B3. Model parameters

B3.1. Commercial fishing selectivity-at-age

The commercial fleet-specific fishing selectivity, S_a^f, is estimated directly for each age from age '*minus*' to age '*plus*'. The estimated decreases from ages *minus*+1 to *minus* and ages *plu*s-1 to *plus* are either assumed to continue exponentially to ages 0 and *m* (maximum age considered) respectively.

Time dependence may be incorporated into these specifications by estimating different selectivity parameters for specific time periods, so that $S_a^f \rightarrow S_{y,a}^f$.

B3.2. Survey fishing selectivity-at-age

For the NEFSC spring and fall surveys, the fishing selectivity, S_a^{surv}, is estimated directly for each age from age 1 to age 8. The selectivity is assumed to remain constant at the level estimated for age 8 for ages 9 and above.

For the NEFSC summer survey, the selectivity is assumed to take the form of an exponential decline up to some maximum age specified, after which it becomes zero:

$$S_a^{surv} = e^{-\lambda(a-1)} \qquad \text{(B31)}$$

The Maine/New Hampshire spring and fall surveys, as well as the Massachusetts inshore surveys are taken as indices of recruitment for the Base Case as their catch-at-length distributions are dominated by lengths corresponding to 1-year-old fish, i.e.:

$$S_a^{surv} = \begin{cases} 1 & \text{for } a = 1 \\ 0 & \text{for } a \neq 1 \end{cases} \qquad \text{(B32)}$$

B3.3. Natural mortality-at-age

$$M_a = 0.2 \qquad \text{(B33)}$$

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